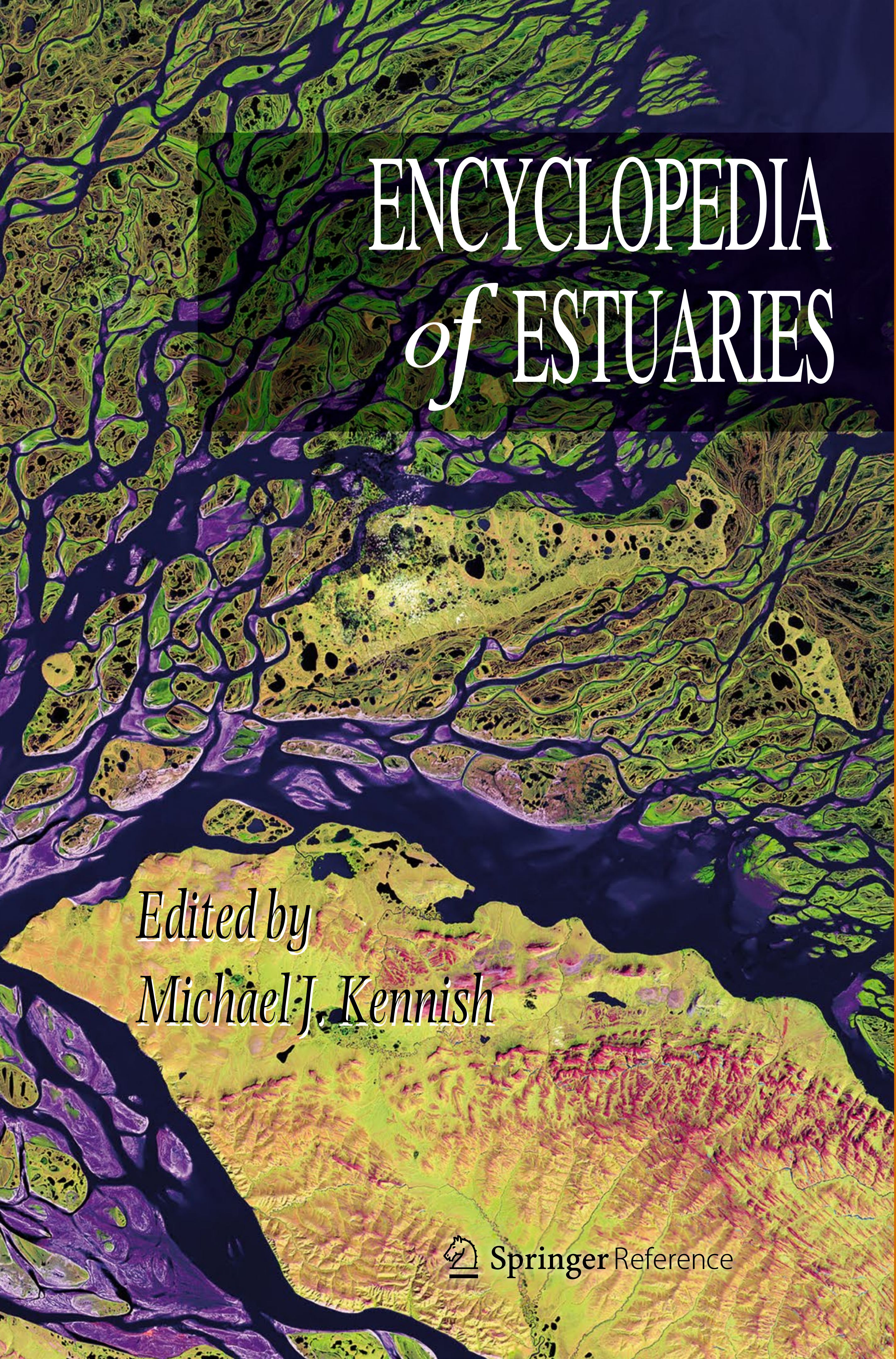

ENCYCLOPEDIA *of* ESTUARIES

Edited by
Michael J. Kennish

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ENCYCLOPEDIA *of* EARTH SCIENCES SERIES

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Encyclopedia of Earth Sciences Series

ENCYCLOPEDIA OF ESTUARIES

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Dr. Michael J. Kennish is a world-class estuarine scientist and a research professor in the Department of Marine and Coastal Sciences at Rutgers University, New Brunswick, New Jersey (USA). In addition to his research on estuaries, Dr. Kennish has conducted biological and geological research on coastal ocean and deep-sea environments. Much of his research has involved the development and application of innovative methods to assess anthropogenic impacts and to determine the condition and health of coastal ecosystems. Dr. Kennish is the author or editor of 14 scholarly books in marine science, and he is the author or co-author of more than 170 articles in science journals and books. In addition, he has edited seven compendium science journal special issues on various topics in marine science.

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Preface

Estuaries are coastal environments with exceptional ecological, recreational, and commercial value. Their diverse habitats provide vital nursery, feeding, and refuge areas for numerous estuarine, marine, and terrestrial organisms. The rich food supply in estuaries supports a multitude of aquatic plant and animal communities. Many marine species of recreational and commercial importance utilize estuaries during at least part of their life history, and therefore these coastal ecosystems play a significant role in the production of marine fisheries. Adjoining wetland habitats are particularly important in the life cycles of fish, shellfish, migratory birds, and other wildlife.

Aside from fisheries, businesses and industries depend on estuaries for successful operation. Most notable in this regard are businesses and industries involving aquaculture, electric power generation, oil and gas recovery, marine biotechnology, tourism, transportation, and shipping. Together, these operations inject hundreds of billions of dollars into the world economy each year. They also employ millions of people worldwide.

Within the land-river-coastal-shelf continuum, estuaries serve a number of critical functions involving the filtering of contaminants, transformation of nutrients, and the biogeochemical cycling of substances. Consequently, they strongly influence the environmental quality of coastal waters. Estuaries also protect coastal watersheds, buffering infrastructure from the damaging effects of storms, floods, waves, and erosion. Because billions of people inhabit coastal areas, these buffering effects are extremely important for the sustainability of coastal communities. Estuaries and adjoining coastal areas are more vulnerable than ever to climate change effects, notably major storms and storm surges, sea-level rise, inundation and flooding, and other hazardous conditions.

Estuaries rank among the most heavily impacted aquatic ecosystems on earth, being affected by a wide range of anthropogenic activities both in adjoining coastal watersheds and on the water bodies themselves. Most of these activities can be linked to rapid population growth and overdevelopment of the coastal zone. While anthropogenic stressors have received the greatest attention of estuarine and marine scientists, some natural stochastic events (e.g., earthquakes, major storms, severe winds, and upwelling) also cause environmental perturbations that can have profound consequences. However, some natural events often occur less frequently than many anthropogenic stressors that affect estuarine environments persistently over more protracted time periods. In addition, multiple anthropogenic disturbances create both acute and insidious problems for many estuarine biotic communities and habitats. Major anthropogenic impacts on estuaries that can pose a threat to their ecological integrity and long-term viability include habitat loss and alteration, eutrophication, sewage, overfishing, chemical contaminants, human-altered hydrological regimes, introduced/invasive species, sea-level rise, subsidence, and floatables/debris. These impacts, together with the other subject areas discussed above, are addressed in the *Encyclopedia of Estuaries*.

The *Encyclopedia of Estuaries* is a volume in Springer's *Encyclopedia of Earth Sciences Series*. It provides thorough and authoritative coverage of the physical, chemical, and biological characteristics of estuaries. As such, it is a major reference work for estuarine and marine scientists, educators, and students. Highly respected authors from around the world have contributed to the encyclopedia on such diverse subjects as biotic communities, essential habitats, food webs, fisheries, hydrology, pollution, conservation, and many more.

With a total of more than 260 topic entries, the *Encyclopedia of Estuaries* covers a broad range of estuarine science and also includes additional information on contiguous environments. It is designed to fulfill the multidisciplinary interests of contemporary estuarine scientists. Although the coverage is not exhaustive on all subject areas of estuaries, the encyclopedia contains detailed information on the topic entries comprising the volume.

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My special thanks go to the authors of the 269 topic entries comprising the *Encyclopedia of Estuaries*. These contributions constitute a tremendous amount of work by many outstanding scientists from around the world. I also acknowledge the Editorial Board assembled for this work consisting of world-class scientists, notably Drs. Daniel Alongi, Donald Boesch, Walter Boynton, Carlos Duarte, Paul Harrison, Robert Howarth, Lisa Levin, and the late Scott Nixon.

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Dr. Charles Finkl, Series Editor for the *Encyclopedia of Earth Science Series*, was instrumental in ensuring the successful completion of the encyclopedia. He continually provided the essential guidance to deal with the many pitfalls encountered during the road to publication. I cannot overemphasize Charlie's important role in the process.

Finally, I thank my colleagues in the Department of Marine and Coastal Sciences at Rutgers University who provided a great source of information during the development of the topic entries for the *Encyclopedia of Estuaries*.

A

ADAPTIVE GOVERNANCE

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Definition

Adaptive governance integrates scientific and other types of knowledge into policies to advance the common interest in particular contexts through open decision-making structures. It is considered as an alternative to and a reaction to the predominant approach to natural resource management in the twentieth century, often referred to as "scientific management." Scientific management emphasized the isolation of a single best practice through rational consideration of the problem and technical expertise, often times developed and implemented by few persons or entities, in a centralized power context (Brunner and Steelman, 2005).

Adaptive governance is best understood as a pattern of practices that emphasizes the adaptation of policy decisions to a diversity of those people who are affected by the decision(s) in question. Any policy must be flexible enough to react to ongoing experiences occurring on the ground. So that managers have the ability to react to unexpected outcomes, enacted policies should be modest and incremental, rather than broad, sweeping, long lasting, and rigid. Authorities and participants must also plan for and budget for monitoring and evaluation of decisions so that policies can be terminated, modified, or nurtured, as the case may be.

Adaptive governance sees the role of science as important and critical but not sufficient in and of itself. Important policy decisions inherently involve values, not just rational consideration of information; no problem is seen as "technical" only. Therefore, adaptive governance depends on and makes use of bottom-up endeavors such as community-based initiatives so that disparate values are brought into the open and considered as part of any policy decision. Moreover, adaptive governance proceeds with the assumption that decision power is fragmented, not centralized.

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ADAPTIVE MANAGEMENT

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Synonyms

Adaptive resource management

M.J. Kennish (ed.), *Encyclopedia of Estuaries*, DOI 10.1007/978-94-017-8801-4,

Definition

Adaptive management is a decision-making process centered upon learning from the outcomes of management actions. Information from monitoring and research is systematically incorporated into future decisions to improve the effectiveness of management.

Description

Adaptive management (AM) is recommended when resource management decisions must be made despite significant uncertainty. AM programs collect information about a system while implementing management actions and apply that information to reduce uncertainty and improve future decisions. AM frameworks typically include monitoring, research, and conceptual and numerical modeling to collect and organize knowledge about a system and incorporate it into future decisions (Williams et al., 2009).

Structured decision making is recommended for the development and ongoing operation of AM programs. Structured decision making is a process in which problems, objectives, management alternatives, and key uncertainties are systematically identified (Gregory et al., 2012). The process includes projecting the consequences of alternative actions into the future to determine the range of likely outcomes and trade-offs while identifying acceptable levels of risk. As resource management has political, social, and economic considerations, formal engagement with stakeholders in the structured decision making process and the development and implementation of an AM program increases the chances that the program will succeed.

AM approaches can be categorized as active or passive. Practitioners of active approaches seek to optimize learning through management experiments, assigning higher value to management actions that will provide the most useful information. Passive approaches use information gained through monitoring of actions and outcomes; however, they generally do not choose actions based on their potential for increasing knowledge. While passive approaches gather information more slowly, they may be beneficial in cases where experimentation is impractical or impossible. Both approaches follow a cyclical pattern of action and assessment. Such a cycle might consist of (1) a planning phase to develop conceptual models, objectives, and management alternatives; (2) a design phase to choose and develop specific actions; (3) an action implementation and monitoring phase; (4) an assessment phase, in which new information is combined with existing information to evaluate the outcomes of actions relative to the objectives; and (5) an adjustment phase to make changes to actions as necessary to improve outcomes. Assessment and adjustment may apply to the current action, returning the cycle to the design phase. Less frequently, managers may use the assessment and adjustment phases to evaluate the AM program itself and revisit the planning phase.

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Cross-references

Adaptive Governance

AEROBIC ENVIRONMENTS

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Synonyms

Aerial environment

Definition

An aerobic environment is one characterized by the presence of free oxygen (O_2), in contrast to an anaerobic environment which is one devoid of free oxygen (WKU, 2013). Aerobic organisms grow or metabolize only in the presence of molecular oxygen (Mekone and Kandel, 1986; Talaro and Talaro, 1993), such as in the upper few centimeters of estuarine bottom sediments where concentrations of free oxygen are significant and chemically oxidizing processes prevail (EPA, 1990). In this environment, aerobic bacteria readily decompose organic matter, breaking down the organic molecules to simple inorganic constituents (Talaro and Talaro, 1993). These organisms require oxygen as their terminal electron acceptor. Anaerobes (anaerobic bacteria), however, grow or metabolize only in the absence of molecular oxygen, such as in the deeper sediment layers of estuarine and marine environments (EPA, 1990).

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Cross-references

Anaerobic Environments

AGE

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Definition

Age. The time elapsed since a water parcel or constituent particle entered a defined water body (Bolin and Rodhe, 1973; Zimmerman, 1976; Takeoka, 1984).

Essential concepts, applications, and methods of estimation

"Age" is a hydrodynamic transport time scale commonly used to characterize the time elapsed between release of a substance (e.g., a pollutant) into a water body and its arrival at a location of concern (Shen and Haas, 2004). Parcels and particles may be released from origin *regions* as well as from point sources; therefore, in addition to the definition provided above, age has also been defined as the time elapsed since a parcel or particle left the region in which its age is prescribed to be zero (Delhez et al., 1999; Deleersnijder et al., 2001). As the time taken by a parcel since entering a water body to reach location **x**, age is commonly considered the complement to "residence time," if residence time is defined as the time taken by a water parcel originating at **x** to leave the water body; the sum of the two time scales is called "transit time" (Takeoka, 1984; Sheldon and Alber, 2002).

Age is unique to each water parcel, spatially heterogeneous within a water body (Monsen et al., 2002; Banas et al., 2007), dependent on source location (Zimmerman, 1976; de Brye et al., 2012), and time dependent (Delhez et al., 1999). Moreover, given that diffusive processes can cause exchange of particles between fluid parcels as they travel through a water body, a parcel is likely to contain particles of different ages (Deleersnijder et al., 2001). Also important are the facts that (1) water is a mixture of different constituents including pure water, salts, dissolved chemicals, biological and mineral particulates, and chemicals sorbed to particles and (2) age of each constituent varies in space and time (Delhez et al., 1999; Delhez and Wolk, 2013). These different constituents are subject to their own unique production and destruction processes, further altering distributions of particle histories – and therefore ages – within a given water parcel (Deleersnijder et al., 2001).

Age is one of several diagnostic transport time scales that can be estimated to distill the details of estuarine hydrodynamic circulation and exchange and to aid in the understanding of linked physical, biological, and chemical processes (e.g., Banas et al., 2007; Lucas et al., 2009; Delhez and Wolk, 2013). This time scale can be used, for example, to backcast release times for substances detected in particular locations (Delhez and Deleersnijder, 2002), assess locations and times of increased estuary vulnerability to river-derived nutrient inputs (Shen and Haas, 2004), understand spatial variations in larval settlement (Banas and Hickey, 2005), and interpret complex hydrodynamic circulation patterns (Deleersnijder et al., 2001; Andutta et al., 2013). Age of water parcels originating at the water surface ("ventilation age," DeVries and Primeau, 2010) is also commonly used to provide insight into ventilation rates in ocean basins (Haine and Hall, 2002; Mouchet and Deleersnijder, 2008).

In estuaries, variability in age may be influenced by freshwater discharge (de Brye et al., 2012), gravitational circulation and stratification (Shen and Haas, 2004), wind (Andutta et al., 2013), bathymetry (Shen and Haas, 2004), tides (Banas and Hickey, 2005), and bottom friction (Andutta et al., 2013). Age is commonly computed with numerical models, using both traditional Lagrangian-based particle tracking techniques (e.g., Andutta et al., 2013) and Eulerian approaches (e.g., Delhez et al., 1999; de Brye et al., 2012). Models representing a broad range of complexity have been used to assess age, including box (Zimmerman, 1976), one-dimensional (Mouchet and Deleersnijder, 2008; Delhez and Wolk, 2013), two-dimensional (Monsen et al., 2002), and three-dimensional (Shen and Haas, 2004) models. Age may also be assessed in the field using substances such as passive dyes (Kratzer and Biagtan, 1997) and radioactive tracers (Delhez et al., 2003; de Vries and Primeau, 2010; Xu et al., 2013).

Summary

Age is a hydrodynamic transport time scale used to convey the time elapsed since a water parcel or particle was introduced to a defined water body. This time scale may be estimated using field or computational techniques to gain insight into the transport and dynamics of substances such as pollutants that are released into surface waters. Age is also used as an interpretive tool for better understanding complex hydrodynamic flows.

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Residence Time
Timescale

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Synonyms

Airborne laser scanning; Airborne laser swath mapping; Airborne light detection and ranging; Laser altimetry; Laser radar

Definition

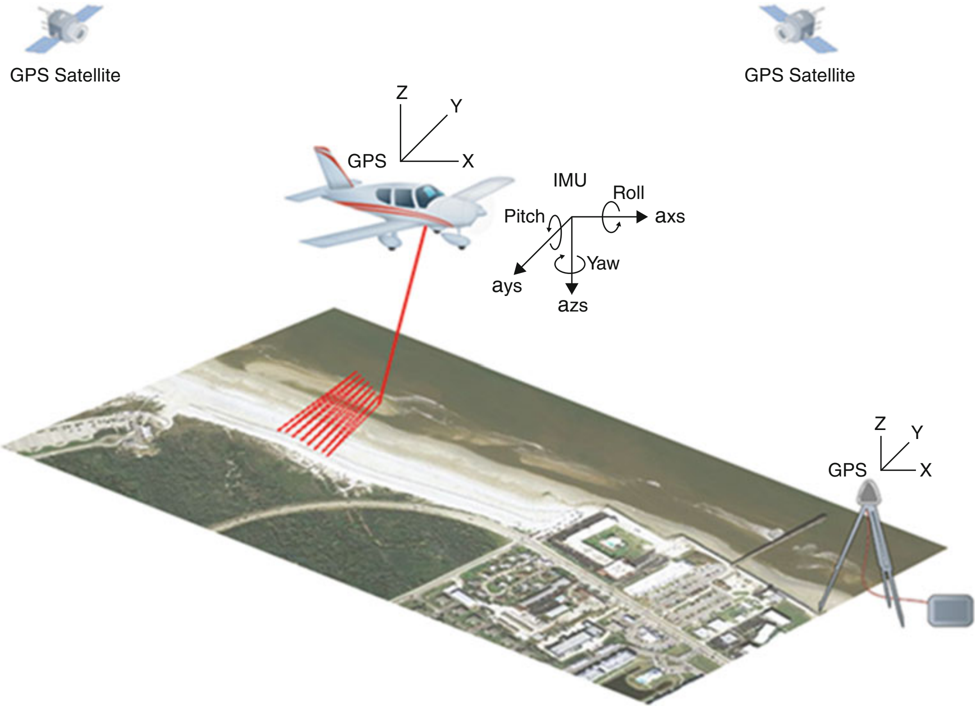
Airborne laser terrain mapping (ALTM) is an active remote sensing technology that employs light detection and ranging (see Light Detection and Ranging (LIDAR)) to measure topography at high spatial resolution over large areas. ALTM pulses a laser to measure the range between an airborne platform and the Earth's surface at many thousands of times per second. Using a rotating mirror or other scanning mechanism inside the laser transmitter, the laser pulses can be made to sweep through an angle, tracing out a line or other patterns on the reflecting surface. With the scan line oriented perpendicular to the direction of flight, it produces a sawtooth pattern of ranges within a strip centered directly along the flight path (Figure 1). An integrated global positioning system (GPS) and inertial navigation unit are used to accurately determine the aircraft position and attitude as each laser pulse leaves the aircraft. This information is then combined with the scan angle and range for each pulse to derive the georeferenced location of the sample points on the reflecting surface (Baltsavias, 1999; Wehr and Lohr, 1999). The result is a densely sampled three-dimensional (3D) point cloud of *x,y,z* values representing the ground and land cover. In addition to spatial information, ALTM systems typically provide a relative measure of the reflection intensity for each surface point based on the return pulse amplitude.

History

ALTM is commonly referred to as airborne lidar mapping. The first airborne lidar systems were developed in the late 1960s as a way to measure height profiles of ice packs and underwater surfaces (bathymetry) where traditional sonar techniques failed due to shallow water depths. The development of profiling bathymetric lidar systems continued through the 1970s. The first system to incorporate a scanning mechanism was the NASA Airborne Oceanographic Lidar (AOL) that became operational in 1977 (Fernandez-Diaz et al., 2013). Terrestrial experiments were conducted starting in 1980 to evaluate the capability of the AOL system to derive topographic maps in areas not suited for photogrammetric methods, such as forested regions. Early results encouraged researchers to develop lidar systems with specific design characteristics targeted for terrestrial applications. However, it was not until the mid-1990s that commercially manufactured units became fully operational (Shan and Toth, 2009). Starting in the 1990s, rapid advancements in enabling technologies such as GPS, IMUs, solid-state lasers, photodetectors, and optical scanners paved the way for the modern-day ALTM system.

System components

ALTM systems consist of three main components. First, the laser ranging unit consists of the laser transmitter, scanner (e.g., oscillating mirror), and a receiver to record the reflected energy. Second, the position and orientation



Airborne Laser Terrain Mapping (ALTM), Figure 1 Example of airborne laser terrain mapping (ALTM) using an oscillating mirror to scan the surface. The GPS on the ground is used to differentially correct the airplane GPS measurements.

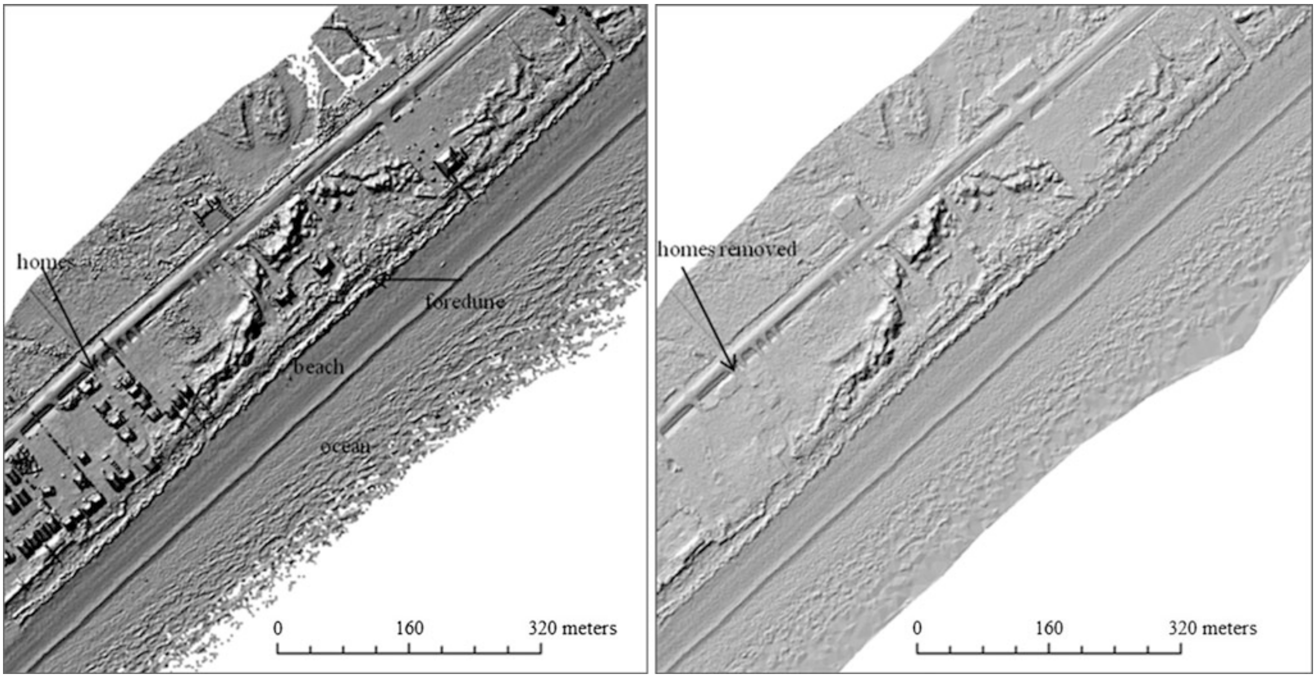
system (POS) consists of a GPS to acquire aircraft positional information and an inertial measurement unit (IMU) to record aircraft orientation (roll, pitch, and yaw). Third, the control and data acquisition unit consists of the onboard hardware and software used to operate the system and record the collected data (Shan and Toth, 2009).

System types

ALTM systems are broadly classified into two groups, those targeted at mapping topography and those targeted at mapping bathymetry. Topographic systems generally operate in the near-infrared wavelengths (e.g., 1,064 nm) of the electromagnetic spectrum that allow for easier compliance with eye safety regulations and where vegetation and other components of the land surface can be highly reflective (Fernandez-Diaz et al., 2013). Bathymetric lidar systems operate in the blue-green range of the electromagnetic spectrum (e.g., 532 nm) to achieve water penetration. However, it should be noted that there are both single- and dual-frequency ALTM systems. For example, some bathymetric systems employ an infrared laser to better detect the water surface and a blue-green laser to map submerged features.

ALTM systems can be further classified into discrete-return and full-waveform systems. Discrete-return ALTM systems typically record multiple returns per an emitted pulse, including a first and last return. Take, for example, a discrete-return system mapping over a forest that can record up to two returns per an emitted pulse. The emitted laser pulse can first interact with the forest canopy resulting in some of the pulse energy being backscattered to the sensor resulting in a measurable return (first return). The remaining pulse energy can subsequently propagate to the land surface below the canopy resulting in a second measurable return (last return). Modern discrete-return systems can record up to four or more returns per an emitted pulse. In contrast, currently available full-waveform systems digitize the entire backscattered laser pulse at very high sampling rates (e.g., 1 GHz). The advantage of this technique is that it provides a much more detailed and complete record of each received signal as it interacts with the landscape. However, this information comes at the expense of increased data storage and increased post-processing requirements. Therefore, full-waveform digitization is not advantageous for all applications, such as those concerned only with mapping the ground surface (Pack et al., 2012).

ALTM systems are generally referred to as small-footprint lidar systems. Small-footprint ALTM systems generate a laser pulse with a smaller beam divergence to reduce the spreading of the pulse as it propagates from the sensor to the surface below. This results in a smaller-diameter laser footprint on the reflective surface, thereby enabling higher-density spatial sampling. The actual diameter of the laser footprint on the surface will be



Airborne Laser Terrain Mapping (ALTM), Figure 2 (*Left*) Shaded-relief image of an ALTM-derived 1-m resolution digital elevation model (DEM) of a section of beach along the Texas coast. (*Right*) Objects, such as homes and vegetation, can be removed through a process called filtering to generate a bare-earth DEM.

a function of the aircraft flying height and the beam divergence. Small-footprint ALTM systems can enable topographic mapping with average spatial resolutions greater than 1 point per meter squared and achievable positional accuracies in the range of 15–30 cm horizontal (x, y) and 5–10 cm vertical (z) (Slatton et al ., 2007).

Data applications

ALTM surveys generate irregularly spaced *x,y,z* point cloud data representing the ground and land cover. The desired end product for many scientific and engineering applications is to derive a bare-earth digital elevation model (DEM) from the data. For multiple return ALTM systems, typically only the last return points are utilized because they have a higher probability of reflecting from the true ground surface. Prior to DEM generation, the point data typically undergo a process called filtering to try and remove non-ground points due to such things as buildings, vegetation, and other occluding objects (Slatton et al., 2007). Many different filtering algorithms have been proposed for ALTM data (e.g., Sithole and Vosselman, 2004). Once the ground points are obtained through filtering, an interpolation method is applied to generate a regularly spaced grid of bare-earth elevations (Figure 2). The achievable spatial resolution of the resultant bare-earth DEMs will depend on the ALTM sampling density and properties of the land cover, but achievable resolutions exceed 1 m. In addition to bare-earth DEMs, the first return points are often used to generate digital surface models (DSMs) of the land cover elevation, such as forest canopy or buildings in urban areas. Furthermore, the ALTM intensity values for each point can be used to derive information about the relative surface reflectance and applied to segment objects captured in the point cloud data.

Summary

ALTM is a well-established mapping solution for large-scale acquisition of topographic elevation data at high spatial resolution. Data derived from such systems have enabled the development of digital elevation models at unprecedented spatial detail. This capability has revolutionized the scientific community's ability to measure land surface dynamics. ALTM system development continues to evolve at a rapid pace including new lidar modalities, such as flash lidar (Pack et al., 2012). This progression in technology will lead to new mapping capabilities and applications.

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Cross-references

Light Detection and Ranging (LIDAR)

ALGAL BLOOMS

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Synonyms

Harmful algal blooms; Nuisance algal blooms; Phytoplankton blooms; Red tides; Spring blooms

Definition

Algal blooms refer to the increase in biomass of algae due to increased growth or due to physical aggregation, leading to an above-normal accumulation and generally adverse ecosystem effects.

Introduction

Algae are a normal part of the aquatic ecosystem; they form the base of the aquatic food web. Of this large and diverse group of organisms, most are microscopic in size, but some are macroscopic. The microscopic algae are most often single cells, but some can form chains or colonies. Most microalgae live in the water column, while others live in or near to the sediment or attached to surfaces for some or all of their life cycle. Macroalgae can be multicellular and complex, the largest of which are the seaweeds.

Algal blooms are an important and natural component of the production of all aquatic systems, but especially those of temperate, subpolar, and coastal waters. Spring blooms are triggered by seasonal warming, increased light availability, water column stratification, and increased nutrient availability from riverine runoff or other sources. These blooms are important for energy and material transport through the food web, and they also play an important role in the vertical flux of material out of the surface waters. These blooms are distinguished from those that are deemed “harmful.” Algae form harmful algal blooms, or HABs, when either they accumulate in massive amounts that alone cause harm to the ecosystem or the composition of the algal community shifts to species that make compounds (including toxins) that disrupt the normal food web or to species that can harm human consumers (Glibert and Pitcher, 2001). HABs are a broad and pervasive problem, affecting estuaries, coasts, and freshwaters throughout the world, with effects on ecosystems and human health, and on economies, when these events occur. This entry focuses on those algal blooms that are HABs. After an introduction to types of HABs and their effects, an emphasis is placed on the ecology and dynamics of the planktonic HABs, their global expansion, and approaches to their prediction and control.

Types of HABs (representative species groups and their effects)

Terminology

The term HAB is an operational one, not a technical one due the diversity of HAB types and effects. Due to the complexity of toxic or ecosystem effects, with the exception of a few species, there are no formal definitions of the concentrations of cells that determine a “bloom.” Many HAB events were formerly referred to as “red tides” because of their pigmentation, but this terminology has been supplanted because not all HABs are red; some may be green, yellow, or brown depending on their specific pigmentation.

While colors are used less frequently to distinguish different types of HABs, toxic properties are gaining favor as a way of distinguishing different types of HABs. By definition, all HABs cause harm, either to ecological, economic, or human health. *Toxic* HABs are those that involve toxins or harmful metabolites, such as toxins linked to wildlife death or human seafood poisonings, as described in more detail below. Of the tens of thousands of algal species, only a few percent have been documented to be toxic, although new toxins are being identified regularly (Landsberg, 2002). Some algal toxins are extremely potent, and thus toxic HABs can occur at cell densities that would not normally be taken to be in “bloom” proportion; they can, for example, sometimes cause poisonings at concentrations as low as a few hundred cells per liter.

Nuisance algal blooms, or NABs, do not produce toxins (or such toxins have not yet been identified), but are able to cause harm through the development of high biomass, leading to foams or scums, the depletion of oxygen as blooms decay, or the destruction of habitat for fish or shellfish such as by shading of submersed vegetation. Another distinction that some investigators have found useful is the group of HAB species that are not necessarily toxic but do cause disruption to the grazer community and thus to trophic transfer and the food web in general. Such blooms have been coined *ecosystem disruptive algal blooms*, EDABs (Sunda et al., 2006).

Adding to the confusing terminology associated with HABs, some HABs are technically not "algae" at all, but rather small animal-like microbes that obtain their nutrition by grazing on other small algae or bacteria; they either do not photosynthesize at all or only do so in conjunction with grazing. These complex and diverse nutritional strategies are described in more detail below. Other "HABs" are more bacteria-like. These are the cyanobacteria (*CyanoHABs*), some of which have the ability to "fix" nitrogen from the atmosphere as their nitrogen source. All of these complexities underscore that the term "HAB" is simply an operational term, recognizing the group of species (not all of which are strictly algae) that can cause harm – to health, to the environment, or to the economy.

Common species groups and harmful properties

Of the thousands of species of marine phytoplankton from hundreds of genera, only a few can be highlighted here. Diatoms are the most common organisms associated with "spring blooms." Diatoms are capable of rapid growth rates. They have a silicate shell and thus have a unique requirement for this element among the algae. Most diatoms are not harmful, but large spring blooms are associated with hypoxia or "dead zones" when the biomass sinks to deeper waters where it decomposes in oxygen-consuming processes. The annual development of summer hypoxia in Chesapeake Bay, for example, is due to these large spring bloom events (Kemp et al., 2005).

Diatoms may also be toxic, as is the case of *Pseudo-nitzschia* spp. that produce *domoic acid* that is responsible for the human illness called *amnesic shellfish poisoning* (Trainer et al., 2012). Exposure to this HAB group is now being linked to seizure and memory loss in laboratory animals and to premature births and strandings in animals such as sea lions (Johnson et al., 2010; Bargu et al., 2012).

Dinoflagellates are among the more common toxic HABs as well as NABs. Dinoflagellates all possess two dissimilar flagella; they often display complex life cycles and typically have much slower growth rates than diatoms. Many produce toxins that can kill fish directly or that intoxicate seafood with toxins that can be passed onto human consumers. One toxic dinoflagellate that causes significant human health effects is *Karenia brevis*, which produces large blooms along the coast of the Gulf of Mexico. This species produces a neurotoxin, brevetoxin, that is responsible for the human illness called *neurotoxic shellfish poisoning* (Backer and McGillicuddy, 2006). Several dinoflagellate species, including *Alexandrium* spp. and *Pyrodinium bahamense* (var. *compressum*), produce saxitoxin, responsible for *Paralytic Shellfish Poisoning* (PSP) (Backer and McGillicuddy, 2006). PSP has been particularly problematic in Southeast Asia, where many human fatalities have been reported over the past several decades (Azanza and Taylor, 2001). Another example of a toxigenic dinoflagellate is *Karlodinium veneficum*. This species has been implicated in fish-kill events in the Chesapeake Bay area (Adolf et al., 2008) as well as in coastal waters of Southwest Africa, Europe, United States, Western Australia, and other temperate coastal environments.

Many HAB dinoflagellates develop large-scale blooms but do not produce toxins that have significant human effects. *Prorocentrum minimum* is one such globally distributed species. This species can produce high biomass blooms that affect the food web by altering ingestion rates and/or growth rates of consumers especially at the larval stages (Heil et al., 2005; Glibert et al., 2008). Benthic species of this genus are known toxin producers, however (Glibert et al., 2012).

Another significant group of HABs is the prymnesiophytes. Many, if not most, *Prymnesium* species are toxic to gill-breathing organisms and thus are responsible for many fish kills around the world, especially in eutrophic waters. For example, *Chrysochromulina polylepis* has been the cause of fish kills along the Norwegian coast, and *Prymnesium parvum* (Carter) is responsible for reoccurring fish kills in coastal and inland waters worldwide (reviewed by Edvardsen and Paasche, 1998; Edvardsen and Imai, 2006; Roelke et al., 2007). Along the North Sea coast, blooms of *Phaeocystis* spp. are common. Because they are mucilage rich, water can turn viscous, and beaches can be drenched in foam from decaying blooms (Lancelot, 1995).

The raphidophytes are yet another common HAB group distributed worldwide. These organisms have often caused large-scale fish mortalities, both in aquaculture settings and in natural environments. Among the more common HAB raphidophytes are *Heterosigma akashiwo* and *Chattonella* spp. The fish-killing properties of these species are due to their production of neurotoxins or to their production of reactive oxygen or other hemolytic agents (Edvardsen and Imai, 2006).

The CyanoHABs are increasingly an important nuisance and toxic HAB group affecting both freshwaters and estuarine and coastal systems worldwide. The most common toxins associated with this group of HABs are hepatotoxins, such as microcystin, nodularin, and cylindrospermopsin, but some species may also produce neurotoxins, such as anatoxin and saxitoxin (O'Neil et al., 2012). The world's largest estuary, the Baltic Sea, is now annually affected by massive CyanoHAB blooms, including species such as *Nodularia*, *Anabaena*, and *Aphanizomenon*. One of the most common HAB types in freshwater as well as in upper estuaries and one exemplifying the effect of both direct and indirect contact is *Microcystis* spp. Exposure to water with this HAB or its toxin can cause skin irritation or respiratory irritation, but prolonged, repeated, or intensive exposure to the HAB toxin has been associated with tumor promotion, especially liver cancer (Backer and McGillicuddy, 2006). As this toxin and other related toxins can affect the nervous system, there has been a suggestion that some neurological diseases such as Parkinson's disease or dementia may also be related to some of the toxic and bioreactive compounds originating from this HAB group (Ibelings and Chorus, 2007). Among the CyanoHAB group are also some species that attach to seagrass, corals, or sediment, such as *Lyngbya* which is also capable of producing a wide array of toxic or potentially toxic compounds, including Lyngbyatoxin and saxitoxin.

Although this entry is focused on planktonic HABs, for the sake of completeness, it is important to mention macroalgal HABs. Macroalgae also dominate the flora of many shallow estuaries, lagoons, and upper embayments, coral reefs, and rocky intertidal/subtidal habitats especially in polluted environments. A well-documented example of such a bloom occurred in 2008, when the macroalgal species *Enteromorpha prolifera* (also called *Ulva prolifera* or sea lettuce) occurred at the venue of the Olympic Games sailing competition, almost blanketing the water with filamentous scum (Hu et al., 2010). Blooms of this magnitude in this region had not previously been observed but have since reoccurred on a near annual basis associated with expanding aquaculture industries and eutrophication. It has been estimated that the cost associated with the management of the *E. prolifera* event in 2008 was greater than \$100 million.

Ecology and dynamics of HABs

For decades, HABs have been studied in all areas of the globe, but there is still much that is not well understood regarding the underlying processes behind the development or outbreak of species or species groups. In the simplest terms, the success of HABs lies at the intersection of their physiological adaptations of the HABs, the environmental conditions, interaction with co-occurring organisms, and physical dynamics of the water body.

Physiological adaptations

In order to grow and make new biomass, algae need to have the necessary materials and energy to make new biomass. The classic paradigm of algal blooms is that of microscopic "plants" or primary producers, dependent on light and the uptake of dissolved nutrients. However, as noted above, many HABs (the diatoms being the primary exception) may both engage in photosynthesis but may also graze on particles, including bacteria, cyanobacteria, other algae, or even bits of fish tissue. This complex nutrition or *mixotrophy* (the mix of different modes of nutrition) imparts advantages to organisms under conditions of low light (when photosynthesis would be reduced), under low nutrients (when inorganic nutrient uptake may be limited), or under conditions of imbalanced nutrient supply. Some species only use mixotrophy as a supplement, while others rely exclusively or nearly so on mixotrophic nutrition (Burkholder et al., 2008). Some of these "algae" in fact do not make their own chlorophyll at all, but rather borrow their chloroplasts from the food they eat, a process termed *kleptochloroplasty*. Most all the major groups of eukaryotic phytoplankton, including most HABs, engage in mixotrophy to some extent (Flynn et al., 2013). Importantly, when an organism undertakes primary production (photosynthesis) and grazing, the two processes provide "more than the sum of the parts" in terms of benefits to the organisms (Mitra and Flynn, 2010), and thus these modes of nutrition have important consequences for understanding HAB success and in modeling the flow of energy and materials in microbial food webs (Flynn et al., 2013).

In addition to the complex nutrition of mixotrophy, the development of specific algal species may be a function of availability of specific nutrient forms. Using sources of nutrients not available to competitors may impart an advantage for certain species or species groups (Glibert and Burkholder, 2011). Many HABs have the ability to utilize organic forms of nitrogen and phosphorus, but mechanisms may vary. Some species have species enzymes for transport or metabolism of certain forms of organic nitrogen or phosphorus, while other species have the capability for the breakdown of organic compounds at the cell surface (Glibert and Legrand, 2006). Yet other species appear to be stimulated when complex organic molecules are provided in conjunction with inorganic nutrients. For example, in mesocosm experiments, Granéli et al. (1985) showed that dinoflagellate populations, including *Prorocentrum minimum*, were stimulated by inorganic nitrogen only when added in combination with humic acids.

Furthermore, the mechanisms for nutrient acquisition and the extent of dependence by a HAB on mixotrophy or on specific dissolved nutrient forms depend not only on the species, but also prevailing environmental factors such as temperature or light (Glibert and Burkholder, 2006). Thus, a given suite of nutrients may have different impacts in different sites and at different times. Smayda (2002) has suggested that different HAB dinoflagellates can be classified into a matrix, based on preferences organized by a nearshore/offshore gradient in decreasing nutrients, reduced mixing, and increasing light. In this matrix, estuarine species are defined as the dinoflagellates that are better adapted to low-light high-nutrient waters, but oceanic species are better adapted to high-light low-nutrient waters.

Many algae, including HABs, also have complex life cycles and behaviors that have important implications for their occurrence, distribution, and dynamics. Among life cycle stages for some algae are benthic cysts or other resting stages that give cells capability to withstand hostile or unfavorable environmental conditions. The metabolic switch from resting stages to motile stages often occurs for a population at or around the same time, allowing actively dividing cells to initiate a bloom. These cysts or spores provide a recurrent seed source or inoculum for planktonic populations, and this characteristic may be a critical factor in determining not only the geographic distribution of species but also their eventual abundance.

Some species form colonial stages during parts of their life cycle that have implications for grazers or may deter or protect species from viral or bacterial infection (Lancelot et al., 2002). Many HAB species are motile, and under certain environmental conditions, their swimming behavior or buoyancy may result in the formation of high-density patches (e.g., Franks, 1992; Kamykowski et al., 1998). Some cyanobacterial species are able to regulate their vertical positioning by synthesis and collapse of gas vesicles. Vertical movement by cells in a stratified environment may help to maximize encounter frequencies for sexual reproduction, minimize grazing losses, and allow cells to obtain nutrients at depth and light at the surface. All of these behaviors have important implications for species and bloom success and serve to underscore the deep complexity of the biology of these seemingly "simple" organisms.

Trophic interactions

The proliferation of HABs reflects the metabolism and specific growth of the "algal" cells, but also the dynamics of the species that are co-occurring with the HAB. It has long been argued that production of allelopathic exudates allows some harmful species to outcompete co-occurring phytoplankton (e.g., Granéli and Johansson, 2003). The ability of a HAB species to build their population under specific conditions is therefore related to the availability of sufficient nutrients to sustain it and to reduction in mortality rates. The response of zooplankton and other grazers to toxic algae is often species specific in terms of behavioral responses and toxin susceptibility. Many HAB species produce more toxins under stress, thereby allowing them to avoid predation and competition by killing their predators and the competing algal species (Granéli and Johansson, 2003). Fish and zooplankton avoid dense concentrations of certain HAB species, and some toxic species are rejected by predators or grazers. Grazing control of HABs can also depend on the population density of the harmful algae, as demonstrated for the brown tides in Narragansett Bay, USA, where suppression of grazing occurs above a threshold concentration (Tracey, 1988). A threshold effect may also occur if the daily production of new harmful cells becomes large enough to saturate the ingestion response of the grazers and the ability of grazers to increase their populations. In that case, population growth can accelerate dramatically (Donaghay, 1988).

Bacteria play an important role in controlling many HABs and regulating their impacts, including their toxicity. Bacteria may also interact with HABs in a positive manner by stimulating their growth. CyanoHABs, in particular, establish mutually beneficial consortia of microorganisms (Paerl and Millie, 1996). A different type of bacterial interaction with HAB species was described by Bates et al. (1995) who showed that the toxicity of the diatom *Pseudo-nitzschia* was dramatically enhanced by the presence of bacteria in laboratory cultures. Likewise, viruses are also now known to have significant impacts on the dynamics of marine communities, and some have been found to infect algae and have been implicated in the demise of red or brown tide blooms (Fuhrman and Suttle, 1993). The extent to which any of the above interactions occur in natural waters and affect HAB dynamics is not well known and represents an important line of inquiry.

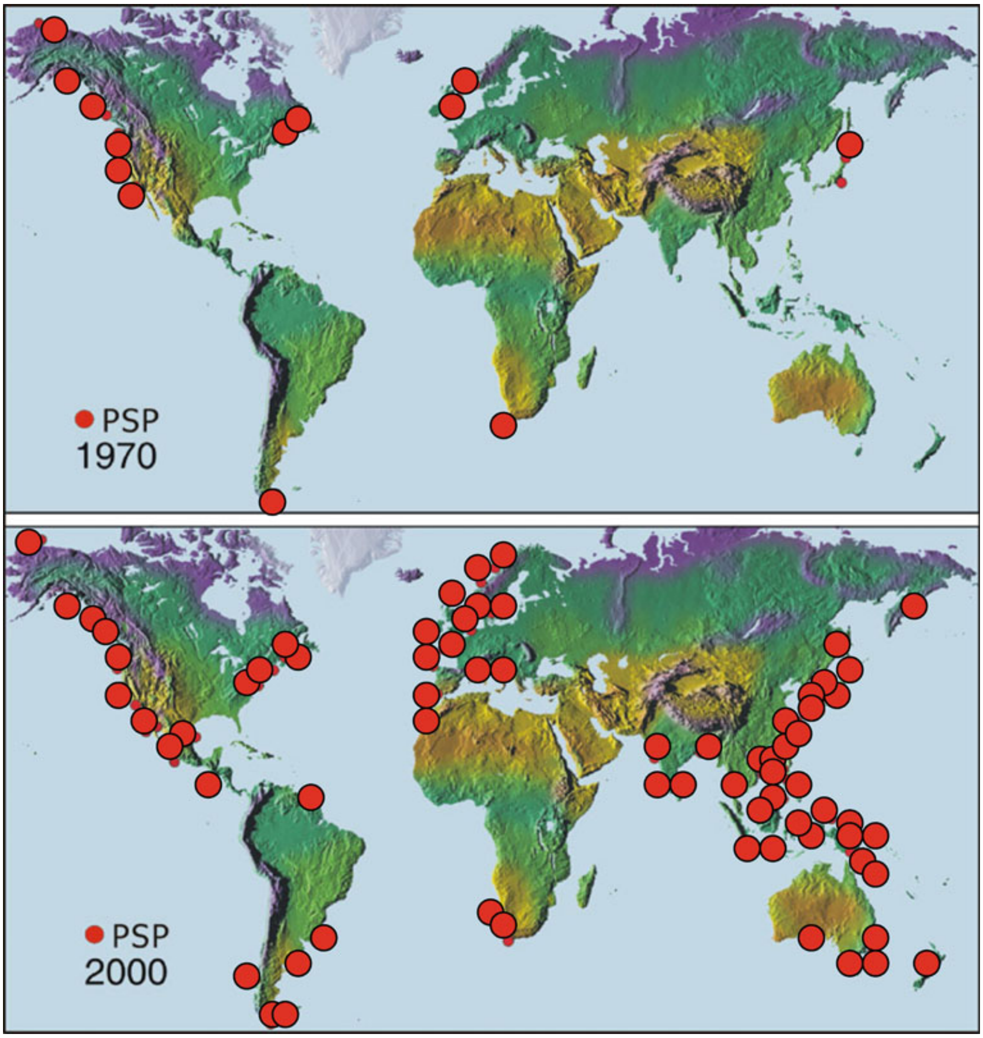
Physical dynamics

The physical environment is also intrinsically intertwined with the biology and ecology of algae. Many large-scale features have direct relevance to HABs or their likelihood for formation, including such features as tidal fronts, coastal jets, and upwelling. Some HABs tend to aggregate subsurface in thin layers. For example, in the Baltic Sea, the heterotrophic dinoflagellate *Dinophysis* may only be found in a 1–2 m layer, but at a depth of 20–25 m (Gisselson et al., 2002). As another example, it has been found off the coast of France that some subsurface layers are comprised of up to 100 % dinoflagellates, most of which are harmful and all of which are mixotrophic, a so-called magic carpet of toxic HABs (Gentien et al., 2008 and references therein). Stratification and mixing have pronounced effects on the distribution and success of HABs. Turbulence, also, has significant consequences for the growth and decline of HABs through its influence on the transport of nutrients, the mixing of phytoplankton through gradients of light, and even through direct impairment of growth. Many questions remain about the specific adaptations of cells to these local environments and the interplay between physics and biology in maintaining these microstructures.

Global expansion of algal blooms, HABs, and their effects

Both toxic and nuisance HABs are increasing throughout much of the world. For example, global occurrences of PSP increased dramatically over the three-decade period from 1970–2000 (Figure 1, Glibert et al., 2005a). Dead zones, another effect of high biomass algal blooms, are also increasing worldwide. The number of dead zones from excessive algal production has doubled each decade since the 1960s. Some systems have shown a progression from episodic to seasonal hypoxia and then, with increased nutrient enrichment, to more and more sustained hypoxia. Dead zones are now found in waters across the globe (Diaz and Rosenberg, 2008).

Although some of the factors contributing to the global expansion are natural, such as biological species dispersal, many others are considered to be a result of human activities, among which nutrient pollution is the most important. The exploitation of natural fish stocks has also, in some cases, led to a decrease in the control of HAB species by removal of the primary grazers through trophic cascade effects. Global climate change may also be important in the increase in HABs.



Algal Blooms, Figure 1 The global expansion of paralytic shellfish poisoning (PSP), one of the many harmful syndromes caused by harmful algal blooms (Figure reproduced from Glibert et al. (2005a) with permission of the Oceanography Society).

Links to eutrophication and altered nutrient stoichiometry

Overenrichment of coastal waters by nutrients is considered a major pollution problem worldwide (Howarth, 2008) and one of the most important factors contributing to global HAB increases (Anderson et al., 2002; Glibert et al., 2005b; Glibert and Burkholder, 2006; Heisler et al., 2008). Nutrient pollution is on the rise because of dramatic increases in human populations in many regions and concomitant increasing demands for energy, increases in nitrogen (N) and phosphorus (P) fertilizer use for agriculture, changes in diet that are leading to more meat production and animal waste, and expanding aquaculture industries (e.g., Howarth, 2008; Glibert et al., 2010; Bouwman et al., 2011; Bouwman et al., 2013). These industries have altered ecosystems through input of feed and feces, only a small percentage of which is incorporated in food biomass.

Increases in total nutrient load can support higher HAB biomass, and alterations in nutrient form can lead to a nutrient regime favoring HAB growth relative to other algal species. At the simplest level, harmful phytoplankton may increase in abundance due to nutrient enrichment, but remain in the same relative fraction of the total phytoplankton biomass. Even though non-HAB species are stimulated proportionately, a modest increase in the abundance of a HAB species can promote noticeable differences in the ecosystem because of its harmful or toxic effects. More frequently, a species or group of species dominates in response to nutrient enrichment or a change in the ratios of nutrient enrichment (Anderson et al., 2002).

The number of examples of algal blooms linked to eutrophication globally is long, and only a few are highlighted here. A species now well documented to be associated with nutrient enrichment is the dinoflagellate *Prorocentrum minimum*. Globally, it is found in regions of the world where the coasts are receiving elevated nutrients from anthropogenic sources (Glibert et al., 2008). In the Chesapeake Bay, blooms of this species now appear to be 10- to 100-fold higher in maximum density than blooms recorded a few decades ago, and these increases track the nutrient load to the bay over the past several decades (Heil et al., 2005). In Puget Sound, Washington, a striking correlation has been found between the growth in documented cases of PSP over four decades and the growth in human population, based on US census statistics, strongly indicative of nutrient loading and eutrophication as the causative agent of change (Trainer et al., 2003). Based on analyses of frustules preserved in cores, blooms of the diatom *Pseudo-nitzschia* spp. in the Gulf of Mexico were also rare prior to the 1950s, but have increased significantly in abundance and frequency since then as nutrient loads from the Mississippi River have risen (Parsons et al., 2002). The Baltic Sea, the Aegean Sea, the Northern Adriatic, and the Black Seas have all experienced increases in HABs coincident with increases in nutrient loading (Granéli et al., 1999; Heil et al., 2005). In Northern European waters, blooms of the mucus-forming HAB species *Phaeocystis globosa* have been shown to be directly related to the excess nitrate content of riverine and coastal waters, that is, the nitrate remaining after other species of algae deplete silicate (Lancelot, 1995). One region where expansion of eutrophication-related HABs has been particularly pronounced has been along the Asian coast, where blooms have expanded in recent years in areal extent (from square kilometers to tens of square kilometers), in duration (days to months), in species, and in harmful impacts (Furuya et al., 2010). These increases all parallel the increase in the use of anthropogenic fertilizers and the accelerated development of China.

On shorter time scales, there are also examples of HABs responding rapidly to injections of nutrients from pulsed events. Beman et al. (2005) reported, during a 5-year study, a strong positive relationship between nitrogen-rich agricultural runoff to the Gulf of California and the development, within days, of extensive phytoplankton blooms. Similarly, *Pseudo-nitzschia pseudodelicatissima* was found a week after elevated ammonium levels were reported in these waters (Trainer et al., 2007), and in Chesapeake Bay, blooms of *Prorocentrum minimum* have been found to follow within days of elevated levels of urea following agricultural applications (Glibert et al., 2001).

Another important consideration in nutrient effects is the relatively recent phenomenon of changing stoichiometry of nutrient supplies. In many parts of the developed world, phosphorus reductions have been undertaken as a means to reduce or control algal blooms (e.g., in sewage effluents and laundry detergents), whereas nitrogen loads often are allowed to remain elevated. Thus, not only have many systems undergone eutrophication, but many are showing signs of reversal due to this single nutrient reduction. The consequence is that many receiving waters are now not only enriched with nutrients, but these nutrients are in proportions that differ from those of decades past – and also diverge considerably from those that have long been associated with phytoplankton growth (Glibert and Burkholder, 2011). Many types of harmful algae appear to be able to thrive when nutrient loads are not in classically defined ideal proportions. Not only are many HABs able to access nutrients not available to competitors through mixotrophy, some species increase toxin production when growing in a state of nutrient imbalance. As examples, toxin production by the flagellates *Prymnesium parvum* and *Chrysochromulina polylepis* increases under both P and N stress, relative to toxin production in more nutrient-balanced growth conditions (Granéli and Flynn, 2006). As another example, in the dinoflagellate *Alexandrium tamarense*, the production of saxitoxin has been shown to increase by three to fourfold under phosphorus deficiency (Granéli and Flynn, 2006).

Adding to the complexity of nutrient effects, in some cases anthropogenic nutrients may not directly stimulate HABs, but may become linked to their growth and abundance following biogeochemical processing or following the stimulation of other components of the food web on which they may depend, or anthropogenic nutrients may be displaced in time and space leading to blooms that are displaced from their nutrient sources. It has recently been found, for example, that *Noctiluca*, the species responsible for classic “red” water, may well be a coastal or offshore manifestation of eutrophication, a mixotroph responding to successional planktonic changes in nutrient availability (Harrison et al., 2011). Relatedly, regulation of single nutrients (e.g., controlling phosphorus without controlling nitrogen) may lead to a situation where an estuary or inshore coastal environment is effectively phosphorus limited and blooms are controlled, but the nitrogen is displaced downstream where it eventually may help to support offshore blooms. Such effects have been documented for the Neuse River Estuary, the mid-region of the Chesapeake Bay, and the southern Baltic Sea region; all of these regions have experienced either episodic or sustained reductions in inshore blooms, but parallel increases in offshore blooms (Glibert et al., 2011 and references therein).

Links to climate

Climate ultimately controls the fundamental parameters regulating algal growth, including water temperature, nutrients, and light, and thus can be expected to result in changes in the species composition, trophic structure, and function of marine ecosystems. Examples can be found through the US and European coasts, where correlations between shifts in HAB species and the timing of their outbreaks and increases in mean water temperature have been documented. In addition, changing greenhouse gases, pH, as well as temperature are related to changes in stratification, vertical exchange, upwelling, precipitation, and related trends, all of which can also influence the habitat for particular HABs (Moore et al., 2008). Moreover, some HABs, especially CyanoHABs, such as *Microcystis* spp., may become more toxic under higher temperatures (Davis et al., 2009).

Climate change may influence HAB expansion, and therefore, the frequency of some blooms may reflect the major changes in ecosystem structure that occur in response to interannual oscillations, such as those related to the El Niño Southern Oscillation (ENSO), or longer term cycles, such as North Atlantic Oscillation (NAO) and the Pacific Decadal Oscillation (PDO). In the northern Iberian Peninsula, for example, the abundance of the harmful dinoflagellate *Gymnodinium catenatum* was high during the mid-1980s when there was a transition from downwelling-favorable conditions to upwelling-favorable conditions following a shift in the NAO index (Alvarez-Salgado et al., 2003). In California, there is circumstantial evidence that a massive domoic acid event in Monterey Bay in 1998 was triggered by post-El Niño runoff (Scholin et al., 2000).

Estuaries: notable examples of sustained HABs in retentive habitats

Estuaries throughout much of the world are sites of frequent HABs (of all forms, including toxic, nuisance, and ecosystem disruptive). Estuaries – whether they be classic river-dominated systems, fjords, coastal embayments, or rias – have many unique features that may be promotive of HABs. Many estuaries are experiencing increasing nutrient loading from pressures of increasing population and housing developments, intensive agriculture in the watershed, and increased aquaculture production. Estuaries may receive considerable riverine input or may be highly retentive with minimal freshwater input or oceanic exchange. The comparatively shallow nature of estuaries (compared to open coasts and offshore regions) means that benthic processes as well as water column processes may be important in providing nutrients for bloom development or maintenance. As a generality, in estuaries and enclosed coastal embayments, exogenous nutrients are often necessary for high biomass blooms to be initiated, but due to long-term buildup of nutrients in estuaries, leading to large sediment reserves of nutrients, recycling and regeneration may sustain blooms at higher densities and for longer periods of time than in years past. Reinforcing feedbacks in estuaries can lead to an acceleration and/or maintenance of eutrophic conditions. For example, increased algal productivity may lead to depressed water column oxygen which, in turn, may result in increased recycling of nitrogen and phosphorus by changes in redox potential, or pH (Kemp et al., 2005; Glibert et al., 2011; Gao et al., 2012). These fluxes will then positively reinforce an ecosystems degradation trajectory and may contribute to blooms being sustained for long periods of time.

One such example is the bloom of *Aureoumbra lagunensis* evident in Laguna Madre, Texas, that lasted for approximately 8 years in the 1990s. Intense rains after years of drought led to a sequence of blooms, and benthic regeneration led to sustained suitable ecosystem conditions (e.g., Buskey et al., 2001). Similarly, a bloom of *Synechococcus* was observed in eastern Florida Bay that followed an injection of phosphorus from two apparent sources: high freshwater discharge from Hurricanes Katrina, Rita, and Wilma that impacted south Florida in 2005 and a very high organic loading from a unique situation of road construction that required mulching of significant amounts of mangroves (Madden, 2010). Consequently, chlorophyll concentrations rose and were sustained at levels roughly eightfold higher than pre-bloom levels for up to 18 months, while such an increase in phosphorus concentration was only observed during the initiation stages of the bloom and then declined.

Detection, prediction, mitigation, and control

Rapid advances are being made in the ability to detect HABs and, in some cases, predict their occurrence and potentially reduce their impacts. Rapid detection capabilities have evolved from classic microscopic methods to detection involving specific molecules and genomes. For example, numerous methods have been developed targeting antibodies against cell surface antigens that are specific for a specific HAB or HAB group that can be detected with a fluorescent signal (reviewed by Sellner et al., 2003). Additionally, molecular probes have been developed for many species, targeting the ribosomal RNA genes and/or their transcriptional products. Many taxon-species probes have been developed.

New methods have been advanced for detection of toxins as well, and some are field based. Additionally, much progress has been advanced in the use of remote sensing capabilities, both remote imagery as well as moored packages and arrays that can detect and provide real-time information on species as well as associated chemistry and physical parameters. Such packages include both moored arrays and remotely operated vehicles that can survey areas more efficiently than was possible from classic shipboard approaches (Sellner et al., 2003). A suite of over 50 such probes in the State of Maryland is allowing managers and the public alike to monitor trends in Chesapeake Bay and rapidly respond when conditions warrant (www.eyesonthebay.net). In situ nutrient sensors are also advancing, with capability developing for some organic forms of nutrients as well as inorganic forms, so that relationships between pulses in nutrient delivery and alterations in salinity due to rainfall, for example, are now possible to establish (Glibert et al., 2005b; Glibert et al., 2008).

Models and forecasting of blooms are advancing very rapidly. There are two general types of HAB models that are useful for management applications. The first is the development of models that predict "general likelihood of occurrence" of HAB species, whereas the second is the development of models that include "explicit" predictions of HAB occurrence in time or space. The former is useful for management in application of long-term actions to reduce the likelihood of future occurrences, i.e., prevention. The latter requires more refinement to understand the physics, biology, and chemistry of the environment, but it can be of more value at the local community level. Coupling the knowledge of the biology of the organisms of interest with robust circulation models of the area of interest has allowed real-time forecasts to be possible. An excellent example of this developing skill is the model that has been developed for *Karenia brevis* blooms off the coast of Florida. This model uses satellite imagery together with a regional circulation model, predicted wind fields, and several biological parameters to forecast where blooms may be found in a several-day period. Operational forecasts are now provided to the public for Florida and several other regions of the United States where similar capabilities are advancing (http://tidesandcurrents.noaa.gov/hab/).

Understanding and predicting algal blooms is important, but the ultimate goal is reducing their occurrence or their impacts once they do occur. The most effective strategy for reducing their likelihood is nutrient reduction. The best cited example illustrating the effectiveness of nutrient reduction is from the Seto Inland Sea in Japan. Between 1965 and 1976, the number of "red tide" outbreaks (high biomass blooms) increased sevenfold, in parallel with the increase in industrial production, but in 1973, Japanese authorities instituted the Seto Inland Sea Law to reduce loadings to half of the 1974 levels over a 3-year period. The number of red tides began to decrease in 1977, eventually falling to less than 30 % of the peak frequency, which had been in excess of 300 blooms per year (reviewed by Imai et al., 2006).

Bloom control strategies may also take the form of mechanical control, the use of filters or booms to remove or exclude cells from certain areas and use of chemical compounds to kill or inhibit bloom cells, or biological control, the use of organisms or pathogens that can lyse, kill, or remove the HABs. Some efforts are finding success with the use of clays that flocculate and remove the HAB cells from the area. Use of clays is well advanced in Korean waters where clay application is used to protect fish cages when HABs develop (Kim, 2006). Overall, however, all of these control measures may have uncertain or unknown environmental impacts, and all such approaches are in early stages of research and development.

Summary and conclusions

In sum, while some algal blooms, such as spring blooms, are natural characteristics of many temperate and coastal waters, and critically important for food webs of marine and freshwater ecosystems, HABs are, in large part, a consequence of anthropogenic activities. HABs are increasing in frequency, magnitude, and ecological and economic effects throughout the world. Understanding of toxins, human health impacts, and the socioeconomic consequences of these blooms – to fisheries and economies – is emerging, but many questions remain unanswered, particularly as new toxins are discovered or characterized. One of the most significant factors contributing to their expansion – particularly in estuaries – is increased nutrient loading from sewage effluent, agriculture, animal operations, and aquaculture. What is clear is that the historic view of phytoplankton responses to eutrophication – increased nutrients promotes increased chlorophyll and high biomass blooms, leading to oxygen deduction and losses in habitat (e.g., Cloern, 2001) – is too simplistic for understanding how HABs respond to the major changes in nutrient loads, forms, and stoichiometry that many systems are now sustaining. Nutrient form and proportion matter and many HABs have physiological mechanisms that enable them to thrive in these environments that are being dramatically altered by human influence. Furthermore, the interplay of biology and physics is only understood at a limited scale, with much to be learned about local scales, microstructures, as well as mesoscale features. Climate changes are adding additional factors that may enhance the likelihood for blooms, and the complexity of ecosystem changes with climate changes means that much is yet to be learned about the direct and indirect effects of climate on HABs. New technologies are advancing toward improved monitoring and prediction, but many such technologies are sophisticated and expensive. Although considerable advances have been made in understanding the biology of HABs, and their interactions with other members of the community at all levels of the food web, there is still much to be learned about how and why specific species respond to specific conditions.

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Cross-references

Ecological Stoichiometry
Eutrophication

ALKALINITY

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Definition

Alkalinity is the name given to the quantitative capacity of water to neutralize an acid to the equivalence point of carbonate or bicarbonate (Water Quality Association, 2000). The total alkalinity of sea water was defined by Dickson (1981) as "... the number of moles of hydrogen ion equivalent to the excess of proton acceptors (bases formed from weak acids with a dissociation constant $K \leq 10^{-4.5}$ at 25 °C and zero ionic strength) over proton donors (acids with $K > 10^{-4.5}$) in 1 kg of sample." For the compounds found in water, the total alkalinity (A_T) is expressed as:

$$\begin{aligned} A_T = {} & [HCO_3{}^-] + 2\left[CO_3{}^{-2}\right] + \left[B(OH)_4{}^-\right] + [OH^-] \\ & + \left[HPO_4{}^{-2}\right] + 2\left[PO_4{}^{-3}\right] + [H_3SiO_4{}^-] + [NH_3] \\ & + [HS^-] - [H^+]_F - [HSO_4{}^-] - [HF] - [H_3PO_4] \end{aligned}$$

where $[H^+]_F$ is the free concentration of the hydrogen ion (Dickson, 2010).

In natural waters, carbonate alkalinity, $A_C = [HCO_3{}^-] + 2[CO_3{}^{-2}]$, tends to comprise most of the A_T due to the common occurrence and dissolution of carbonate rocks and presence of carbon dioxide in the atmosphere. Other common natural components of A_T are borate, hydroxide, phosphate, silicate, nitrate, ammonia, sulfide, and the conjugate bases of some organic acids. In anoxic conditions the relative role of sulfide, ammonia, and phosphate components of A_T increases (Volkov et al., 1998). In coastal regions, especially estuaries, dissolved organic matter can significantly contribute to A_T (Kim and Lee, 2009).

Alkalinity can be measured by titrating a sample with a strong acid until all the buffering capacity of the aforementioned ions above the pH of bicarbonate or carbonate is consumed (i.e., total titratable alkalinity). This point is functionally set to pH 4.5. At this point, all the bases of interest have been protonated to the zero level species; hence, they no longer cause alkalinity.

An addition (or removal) of CO_2 to a solution does not change the alkalinity. Addition of CO_2 to a solution in contact with a solid can affect the alkalinity, especially for carbonate minerals in contact with groundwater or seawater. The dissolution (or precipitation) of carbonate rock has a strong influence on alkalinity. In open ocean waters, alkalinity can be connected with salinity and temperature with a functional dependence (Lee et al., 2006). Rivers can act as either a source or a sink of alkalinity.

The actual units for the alkalinity titration are moles or equivalents per volume ($mol\ L^{-1}$ or $Eq\ L^{-1}$). They can be converted to $mol\ kg^{-1}$ or, in terms of calcium carbonate, to $mg\ CaCO_3\ L^{-1}$.

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Cross-references

pH
Water Quality

AMPHIPODS

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Synonyms

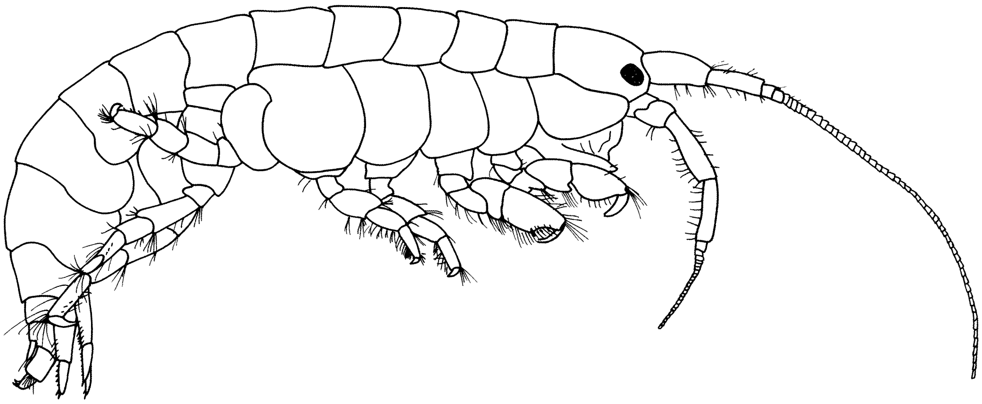
Beach fleas; Beach hoppers; Sand fleas; Sand hoppers; Side swimmers; Scuds; Skeleton shrimps

Definition

Amphipods are crustaceans from the order Amphipoda (Arthropoda, Crustacea, Malacostraca, Peracarida).

Amphipods are a diverse group of small crustaceans that are important members of most aquatic communities. They are mostly small (less than 2 cm) and laterally flattened with no carapace, a thoracic region with seven pairs of pereopods (walking legs) and an abdominal region bearing three pairs of pleopods and three pairs of uropods (Figure 1). The sexes are separate and frequently dimorphic, with males having an enlarged gnathopod involved in mate guarding. Following mating, their eggs are brooded within a marsupium and develop directly after hatching.

Amphipods are highly abundant in most estuarine sediments (up to 100,000 individuals per square meter),



Amphipods, Figure 1 Grazing amphipods from the family Ampithoidae are abundant in estuarine seagrass beds.

occurring as members of the epibenthic community or as burrowing infauna. They are one of the most conspicuous members of the invertebrate fauna inhabiting sandy or muddy beaches where they burrow into sediments or are closely associated with seagrass and algal wrack. Amphipods are abundant as fouling organisms on hard substrates and often numerically dominate the epifauna of seagrasses and macroalgae. Many species are also commensal with benthic invertebrates including sponges, ascidians, and bryozoans. Relatively few species are exclusively planktonic. While they have no larval stage that disperses in the water column, benthic amphipods are often highly mobile and colonize new habitats quickly.

Amphipods display a great diversity of feeding strategies. They include detritivores, suspension feeders, scavengers, predators, herbivores, and parasites (rarely). Those that process sediments play an important role as bioturbators in estuaries, altering the physical and chemical conditions, and thus nutrient fluxes, of the sediments. Grazing amphipods in vegetated habitats can alter the composition of primary producers and play an important role in limiting the abundance of epiphytes that can negatively affect seagrass health. Amphipods frequently comprise a large component of the diets of predatory fish and birds and thus play an important role in the transfer of energy from primary producers to higher trophic levels.

Due to their short generation times, direct development, ease of culture, and intimate associations with sediments, amphipods are routinely used in ecotoxicology as test organisms for evaluating sediment and water quality and as biomonitors for environmental contamination.

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ANADROMOUS

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Definition

Organisms such as fishes which hatch in fresh water and migrate to higher salinities such as the sea to mature and then migrate back into fresh water to spawn.

Background

Originating in the nineteenth-century Russian literature and refined from the original term ("contranatant"; Meek, 1916; Subnikov, 1976) by Meyers (1949), anadromous is one of the three types of migration between the sea and fresh water (McDowall, 1987). The directed movement by anadromous fish between these markedly different habitats is specifically associated with reproductive phases of their life cycle. Of the approximately 20,000 species of fish around the world (Cohen, 1970), McDowall estimated that 54 % are anadromous. They are most common in northern subpolar and cooler temperate waters.

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ANAEROBIC ENVIRONMENTS

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Synonyms

Non-aerial environment

Definition

An anaerobic environment is characterized by the absence of free oxygen but may contain atomic oxygen bound in compounds such as nitrate (NO_3), nitrite (NO_2), and sulfites (SO_3), in contrast to an aerobic environment where free oxygen is available (WKU, 2013).

Description

Some organisms are successfully adapted to anaerobic environments (EPA, 2012). These organisms do not use O_2 as their electron acceptors. There are a number of substances that they will use as a substitute for O_2, including (among others) nitrate (NO_3), ferric iron ($Fe_3{}^+$), manganese ($Mn_2{}^+$), sulfate ($SO_4{}^{2-}$), and carbon dioxide (CO_2) (Scharf, 2000). Bacteria use sulfate as an acceptor for electrons in place of oxygen under anaerobic conditions. As the reduced organic tissue is oxidized to CO_2, the sulfate is reduced to sulfide (Nietch, 2000).

In bottom waters of stratified estuaries, oxygen consumed primarily by bacteria will exceed oxygen input via atmospheric and photosynthetic reoxygenation. While the anoxia/hypoxia that develops will eliminate almost all estuarine and marine organisms, a large number of bacteria and protists can still remain active in these areas by changing their metabolism to anaerobic respiration (Dang et al., 2008). If oxygen becomes so low that the system becomes anoxic, which occurs frequently in shallow turbid estuaries in the summer time, anaerobic mechanisms for decomposition become important. Carbon turnover occurs continuously in subtidal and intertidal sediments where oxygen is depleted just a few millimeters below the sediment surface. Much of the organic matter undergoes decomposition in an estuarine anaerobic environment via sulfate reduction (Nietch, 2000). Preservation of skeletal remains in these environments appears to be correlated with the abundance of dissolved organic matter rather than with high sedimentation rates (Simon et al., 1994).

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Cross-references

Anaerobic Environments
Anoxia, Hypoxia, and Dead Zones

ANOXIA, HYPOXIA, AND DEAD ZONES

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Synonyms

Anoxic; Anoxic basins; Euxina; Hypoxic, Low dissolved oxygen; Oxygen minimum layer; Oxygen minimum zone

Definition

Anoxia is a condition of no, or at times very little, dissolved oxygen in marine or freshwater systems, which has drastic consequences to normal ecosystem functioning including biogeochemical cycling.

Hypoxia is a condition of low dissolved oxygen concentrations in marine or freshwater systems, which has adverse consequences to normal ecosystem functioning including biogeochemical cycling that range from mild to severe disruption.

Dead zone is an area of hypoxia or anoxia that is related to anthropogenic activity.

Introduction

Oxygen is necessary to sustain the life of fishes and virtually all higher invertebrates. When the supply of oxygen is cut off from bottom waters, usually from temperature and/or salinity stratification of the water column that separates surface and bottom layers, or consumption of oxygen through respiration exceeds resupply, oxygen concentrations can decline below levels that will sustain animal life. This condition of low oxygen is known as hypoxia. Water devoid of oxygen is referred to as anoxic. Ecologists have borrowed the term hypoxia from the medical community, but the meaning and processes for the environment are the same. The medical condition is a deficiency in the amount of oxygen reaching tissues. Similarly, a water body can be deprived of adequate oxygen for proper ecosystem functioning.

Hypoxic areas are sometimes referred to as dead zones. A term first applied to the northern Gulf of Mexico hypoxic area, which is related to excess agricultural and municipal nutrients discharged from the Mississippi and Atchafalaya Rivers (Rabalais et al., 2002, Rabalais et al., 2010), and refers to the fact that fish and shrimp avoid and migrate out of hypoxic areas. When fishermen trawl in these zones, little to nothing is caught. The term dead zone is best applied to coastal waters affected by anthropogenic activities where oxygen depletion occurs in otherwise normoxic (well oxygenated) waters and not to naturally occurring oceanic oxygen minimum zones.

The migration of mobile fishes and invertebrates can be obvious, and mortality of smaller sessile invertebrates can be overlooked, but in the absence of larger fauna, smaller species (e.g., foraminiferans and nematodes) and microbes persist and thrive, such that hypoxic areas are not truly dead or devoid of life. In addition, the fully oxygenated upper water column continues to support diverse communities, including productive fisheries. Globally, there are four basic types of low-oxygen marine environments:

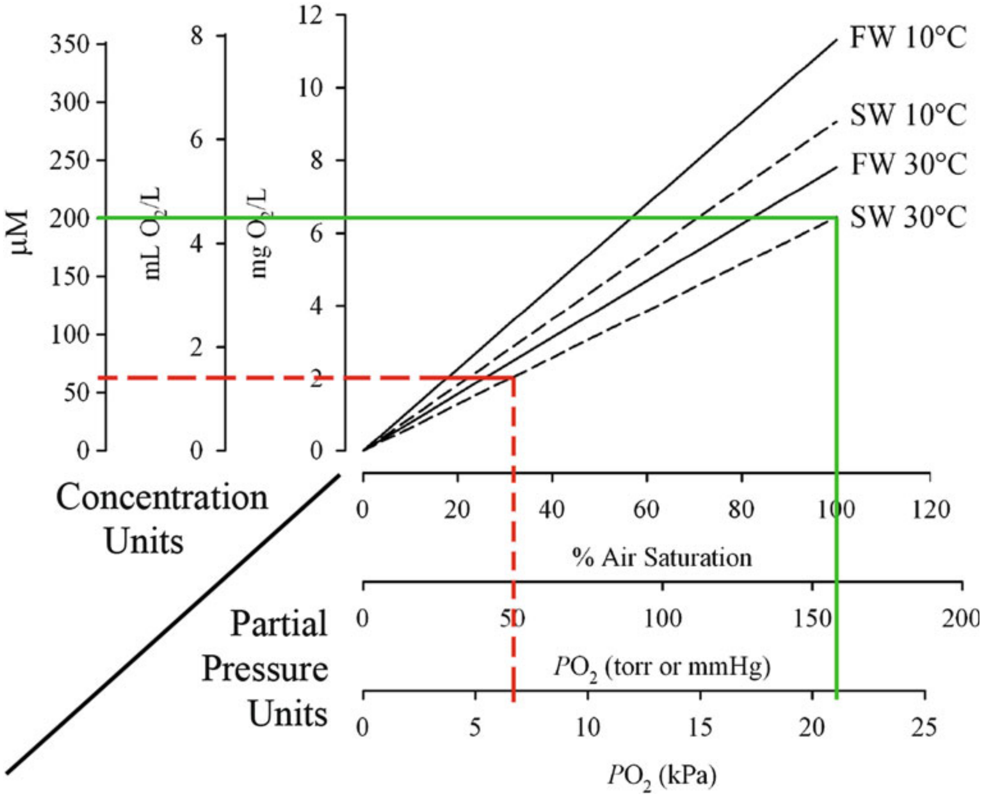
- Naturally occurring oceanic oxygen minimum zones
- Naturally occurring coastal upwellings
- Naturally occurring anoxic deep basins
- Anthropogenic-related coastal, estuarine, and brackish water hypoxia

Dead zones are just one of many consequences of anthropogenic activities (MA, 2005). Human population is expanding exponentially, recently passing six billion, and will likely exceed 8-10 billion by the year 2050. This expansion has lead to an exponential modification of landscapes at the expense of ecosystem function and services including pervasive effects from fuelling coastal primary production with excess nutrients, the primary factor leading to formation of hypoxia, to fishing down the food web (Vitousek et al., 1997; Jackson et al., 2001; Foley et al., 2005; Lotze et al., 2006). Long-term records of nutrient discharges provide compelling evidence of a rapid increase in the fertility of many temperate coastal ecosystems starting about 50 years ago (Galloway et al., 2004; Galloway et al., 2013). On a global basis, by 2050, coastal marine systems are expected to experience, from today's levels, a 2.4-fold increase in nitrogen and 2.7-fold increase in phosphorus loading from this population expansion (Tilman et al., 2001), with serious consequences to ecosystem structure and function. The question asked by Foley et al. (2005) is as follows: are land-based activities degrading the global environment in ways that undermine ecosystem services, which in turn undermine human welfare? When it comes to dissolved oxygen and the development of dead zones, the answer is yes.

Dissolved oxygen concentrations

While many authors and water quality regulations focus on concentrations of oxygen below 2–3 mg O_2/L (see Figure 1 for conversions to other units) as a threshold value for hypoxia in marine and brackish water environments, such arbitrary limits may be unsuitable when examining potential impacts of hypoxia on any one given species (Vaquer-Sonyer and Duarte, 2008). Hypoxia becomes detrimental when behavioral and physiological responses result in altered behavior or negative impacts, such as reduced growth, loss of reproductive capacity, mortality, reduced biodiversity, loss of secondary production, and stressed fisheries. For example, Atlantic cod (*Gadus morhua*) growth in St. Laurence is reduced below about 7 mg O_2/L or 70 % air saturation (Chabot and Dutil, 1999). Shrimp and fish avoid dissolved oxygen below 2 mg O_2/L (approximately 30 % air saturation) in the northern Gulf of Mexico hypoxic zone, while sharks and rays emigrate from the area at oxygen concentrations below 3 mg O_2/L (Rabalais et al., 2001).

Because of the low solubility of oxygen in water, small changes in the absolute amount of oxygen dissolved in water lead to large differences in percent air saturation. For freshwater at 20 °C, 9.1 mg of oxygen (O_2) will dissolve in a liter of water, so a 1 mg O_2/l drop is about an 11 % decline in saturation. In addition, oxygen solubility is strongly dependent on temperature and the amount of salt dissolved in the water. Saturation declines about 1 mg O_2/L from 20 to 26 °C and about 2 mg O_2/L from freshwater to seawater at similar temperatures (Benson and Krausse, 1984). Therefore, depending on temperature and salinity, water contains 20–40 times less oxygen by volume and diffuses about 10,000 times more slowly through water than air (Graham, 1990). Thus, what appear to be small changes in oxygen can have major consequences to animals living in an oxygen-limited milieu (Rabalais and Gilbert, 2009). Physiologically, higher temperatures also increase metabolic requirements for oxygen



Anoxia, Hypoxia, And Dead Zones, Figure 1 Nomogram for dissolved oxygen in freshwater (*FW*) and seawater (*SW*) at 10 °C and 30 °C (Modified from Diaz and Breitburg, 2009). Concentration units are on y-axis, and partial pressure units are on x-axis. *Red dotted line* is 2 mg O_2/L and *green solid line* is 100 % solubility of oxygen in seawater at 30 °C.

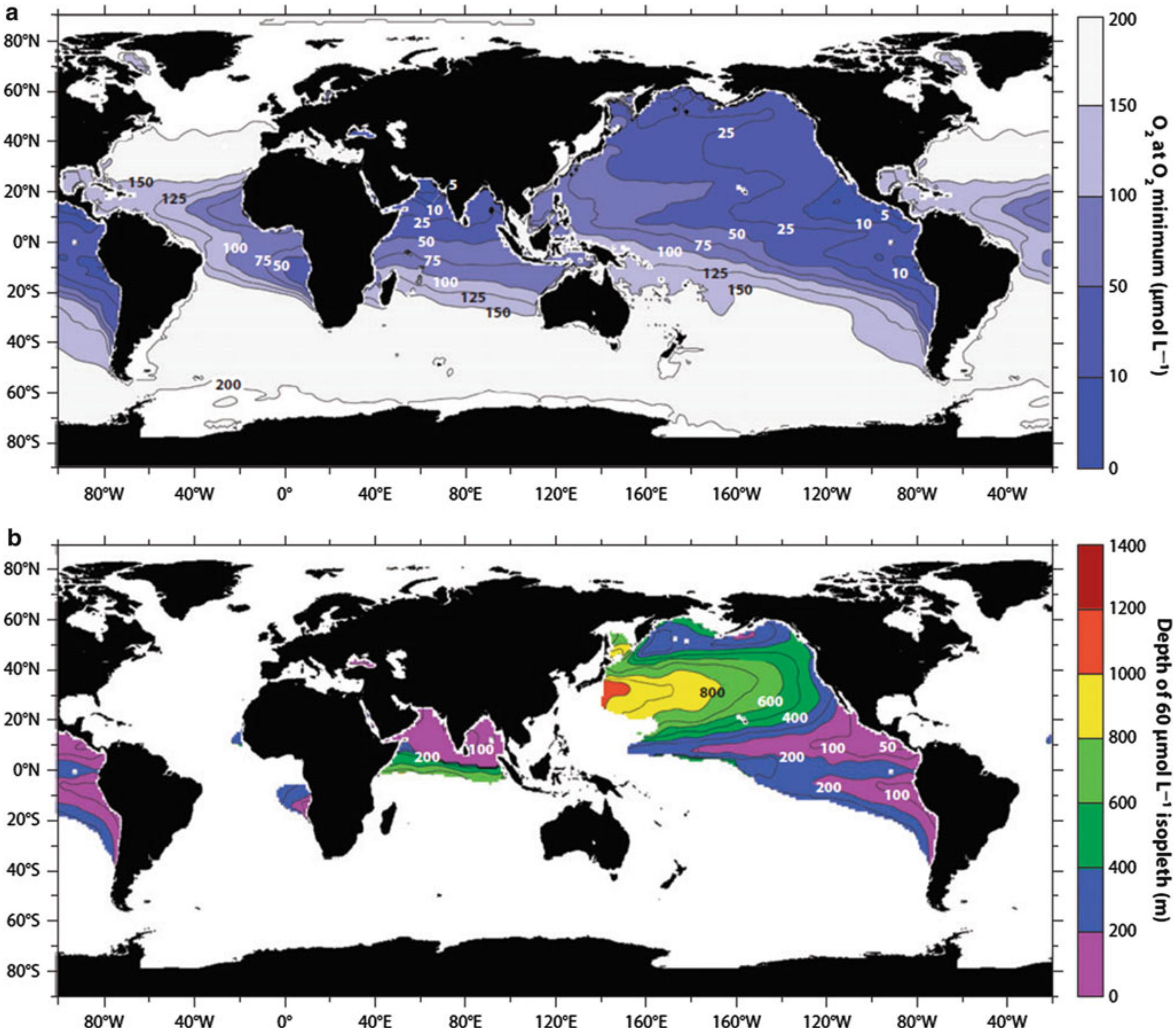
and increase rates of microbial respirations and, therefore, oxygen consumption. For salmonid fishes, oxygen can become limiting at relatively high values, and even air saturation can be limiting at higher temperatures (Fry, 1971). Concentrations of dissolved oxygen below 2-3 mg O_2/L are a general threshold value for hypoxia for marine and estuarine organisms and 5-6 mg O_2/L in freshwater. However, species and life stages differ greatly in their basic oxygen requirements and tolerances (Vaquer-Sonyer and Duarte, 2008).

Naturally low dissolved oxygen environments

By any definition, oceanic oxygen minimum zones (OMZs) are the largest low dissolved oxygen areas on earth and cover about 30 million square kilometers of open ocean (Figure 2), much of which is not near the continents (Paulmier and Ruiz-Pino, 2009). Globally, where OMZs contact the bottom, about 1,148,000 km^2 of continental margin seafloor is estimated to have bottom water oxygen concentrations <0.7 mg O_2/L (Helly and Levin, 2004). The principal factors that lead to the formation of OMZs are high surface productivity, old water mass age, and limited circulation. In addition, upwelling associated with continental margins leads to higher productivity and greater oxygen demand and also contributes to a thicker OMZ and lower oxygen concentrations (Helly and Levin, 2004).

Upwelling areas can also develop extensive hypoxia as deepwater nutrients are added to surface waters increasing production that eventually sinks and decomposes. Hypoxia associated with upwelling is not as long-lived and stable as that associated with OMZs but can periodically reoccur. The best known upwelling is along the coast of Peru and Chile associated with El Niño. Hypoxia associated with this type of coastal upwelling is not as long-lived and stable as that associated with OMZs. Upwelling can interact strongly with low-oxygen water masses to produce intense shelf hypoxia; this is observed off of Oregon, USA (Grantham et al., 2004), and Chile (Fuenzalida et al., 2009), Africa (Monteiro et al., 2008), and India (Naqvi et al., 2000).

Upwelling associated with both the Humboldt (Escribano et al., 2004) and Benguela Current systems (Chapman and Shannon, 1985; Monteiro et al., 2008) develops extensive severe hypoxia and anoxia that adversely affect pelagic and benthic species (Arntz et al., 2006). For example, in 1994, persistent and pronounced hypoxic conditions developed off the coast of central and northern Namibia over much of the continental shelf. These conditions displaced



Anoxia, Hypoxia, And Dead Zones, Figure 2 General global distribution of oxygen minimum zones (60 μmol/L). (**a**) *Colors* indicate oxygen concentrations at the depth of minimum oxygen. (**b**) Depth of the 60 μmol/L isoline in meters (From Keeling et al., 2010).

juvenile Cape hake (*Merluccius capensis*) offshore from their typical inshore habitat, subjecting them to heavy mortalities from predation by larger hake and from commercial trawling (Hamukuaya et al., 1998). Lobsters (*Jasus lalandii*) were trapped by low-oxygen waters advecting onshore that caused a "walkout" and mass mortality (Cockroft, 2001).

Today, there are a number of isolated ocean basins that are hypoxic/anoxic due to restricted water circulation combined with high primary production in surface waters. Among them are the permanently anoxic Black Sea and Cariaco Basin and several smaller basins along the California, USA, coast (Santa Monica, Santa Barbara, San Pedro, San Nicolas basins) that are reoxygenated periodically (Berelson, 1991). Some deep fjords that are permanently anoxic, such as the Framvaren, Norway, have a similar biogeochemistry to oceanic anoxic basins (Skei, 1983). But, most fjords have some seasonal deepwater renewal, which keeps them from being anoxic all year.

In the case of the Black Sea, anoxic conditions occur beneath 100–200 m of oxygenated surface waters in the open sea area. The restricted connection with the Mediterranean Sea, strong stratification, and geomorphology make the Black Sea a sink for nutrients and other materials (Richards, 1965; McQuatters-Gollop et al., 2008). The persistent anoxic basin in the open Black Sea is the largest naturally occurring anoxic area on earth. It covers about 75 % of the basin's area and is distinct from and not related to the eutrophication-related seasonal hypoxia that has occurred on the shallow northwestern continental shelf (Tolmazin, 1985; Zaitsev, 1992; Mee, 2001, 2006). The Cariaco Basin also supports a large persistent anoxic area below 250 m. Restricted circulation and high primary production within the basin support this anoxia (Müller-Karger et al., 2001; Müller-Karger et al., 2004). This naturally occurring anoxic basin allows for sediments to be deposited without bioturbation, forming varves of alternating light and dark color, which correspond to the dry or rainy season (Haug et al., 2001). Because of water column anoxia, anoxic basins have a unique biogeochemistry that resembles that of anaerobic sediments (Madrid et al., 2001; Stoeck et al., 2003). Bacteria inhabit both the oxic and anoxic portions of the water column, with a maximum in the suboxic interface (Taylor et al., 2001). The suboxic layer oscillates between 200 and 300 m in the Cariaco Basin and 150 and 200 m in the Black Sea (Oguz, 2005).

Recently, a troubling decline in the oxygen content in the open ocean is becoming apparent. Much of the decline is related to global climate change but declining trends near the coastline may be stronger and related to human activities (Stramma et al., 2008; Stramma et al., 2012; Gilbert et al., 2009; Keeling et al., 2010). The upper depth limit of OMZs has major implications for fisheries. Expansion of OMZs toward the surface in the eastern tropical Pacific has limited the depth distribution of tropical pelagic marlins, sailfish, and tunas into a narrower surface layer of oxic water about 50-100 m thick (Prince and Goodyear, 2006). The high-performance physiology of these fishes leads to a relatively high hypoxic threshold (Brill, 1996), making any reduction in oxygen problematic. Declining oxygen and expansion of the OMZ in the tropical northeast Atlantic Ocean toward the surface are also restricting usable habitat of billfishes and tunas. From 1960 to 2010, Stramma et al. (2012) found hypoxia-based habitat compression to decreasing their suitable habitat by 15 %. The combination of shallowing of OMZs encroaching onto outer continental shelves and increased coastal eutrophication-induced hypoxia will eventually reduce suitable habitat space for both pelagic and demersal fishes.

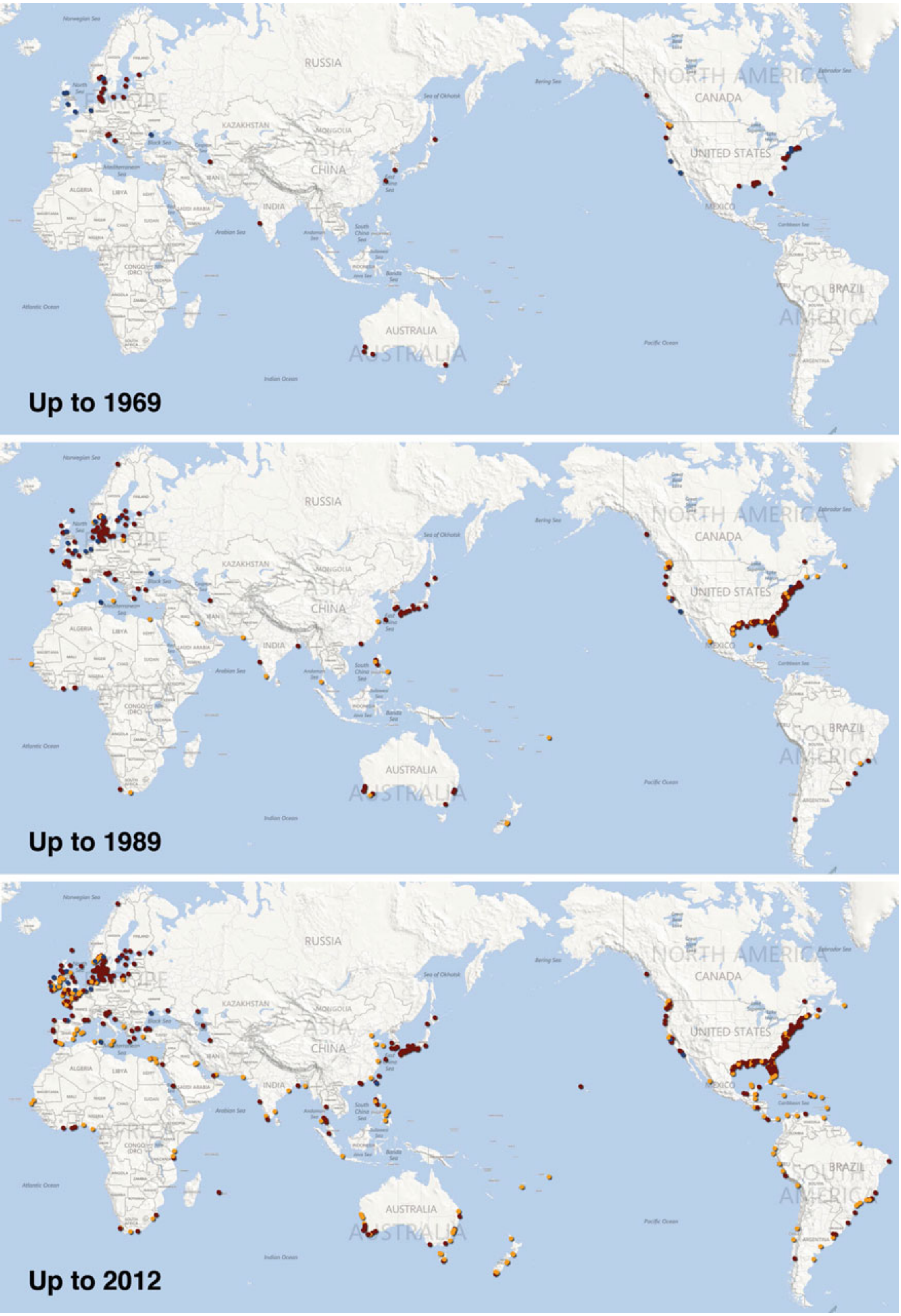
Anthropogenic influence on dissolved oxygen

In contrast to OMZs, upwelling zones, and deep basins, much of the hypoxia and anoxia in shallow coastal marine areas have developed within the last 50 years and are closely associated with anthropogenic activities. Diaz and Rosenberg (1995) noted that no other environmental variable of such ecological importance to estuarine and coastal marine ecosystems has changed so drastically in such a short period of time. Through time, there have been consistent trends of increasing severity in duration, intensity, or extent of hypoxia in areas with long-term data, for example, the northern Adriatic Sea (Justić et al., 1987). Once a system develops hypoxia, it can quickly become an annual event and a prominent feature affecting energy flow (Elmgren, 1989; Baird et al., 2004). Currently, there are over 600 hypoxia areas associated with anthropogenic activities in the world's coastal areas covering more than 245,000 km^2 of sea bottom (Diaz and Rosenberg, 2008; Diaz et al., 2010; Conley et al., 2011; Figure 3).

The worldwide distribution of coastal hypoxia is related to major population centers or is closely associated with developed watersheds that export large quantities of nutrients, specifically nitrogen and phosphorus. Up to 1970, there were scattered reports of hypoxia in North America and northern Europe. By the 1990s, coastal hypoxia was prevalent in North America, northern Europe, and Japan. By the 2000s, there were increased reports of hypoxia in South America, southern Europe, and Australia (Figure 3). Considering the close association of human population and hypoxia, it is likely that Asia and the Indo-Pacific have many unreported hypoxic areas.

Eutrophication can be defined as the increase in the rate of production of organic matter and accumulation of that organic carbon within an ecosystem (Nixon, 1995; Rabalais, 2004). This is typically in excess of what an ecosystem is normally adapted to processing and is only part of a complex web of stressors that interact to shape and direct ecosystem level responses to stressors (Cloern, 2001). The primary driver of eutrophication in both freshwater and marine systems is excess nutrient enrichment, but physical conditions that limit reaeration of bottom water are also necessary for the development of hypoxia. Phosphorus is generally the limiting nutrient in freshwater (Schindler, 1977), and increases in anthropogenic phosphorus have caused increased algal production and eutrophication in freshwater ecosystems worldwide (Carpenter et al., 1999; Smith et al., 2006). For marine systems the limiting nutrient tends to be nitrogen (Howarth and Marino, 2006). This basic difference is related to the physical properties of phosphorus and nitrogen compounds and their biogeochemical cycling through the freshwater and marine environments. Basically, phosphorus tends to be more limiting in fresher, brackish waters and river plumes, and nitrogen is more likely to be limiting in the marine end member of an estuary or a freshwater-dominated coastal system.

Eutrophication and associated hypoxia in freshwater systems became widespread in the twentieth century, but effective nutrient management has reversed this trend where it has been rigorously implemented (Jeppesen et al., 2005). In tidal portions of rivers and other water bodies near dense population centers, severe hypoxia and anoxia have been caused by discharge of raw sewage, which is high in both nutrients and organic matter. Areas devoid of fishes were reported as early as the late 1800s, for example, the Mersey Estuary, UK, and persisted until improvements in sewage treatment were implemented (Jones, 2006). Much of the hypoxia and anoxia in shallow coastal marine and estuarine areas are recent in origin and related to a combination of nitrogen and phosphorus from agriculture and human waste and atmospheric deposition of nitrogen.



Anoxia, Hypoxia, And Dead Zones, Figure 3 (Continued)

Annual hypoxia does not appear to be a natural condition for marine waters except for those systems previously described. Even in Chesapeake Bay, which had hypoxia when oxygen measurements were first made in the 1910s in the Potomac River (Sale and Skinner, 1917) and 1930s in the main stem channel (Officer et al., 1984), the geological record suggests that low oxygen was not an annual, seasonally persistent feature of the system prior to European colonization (Cooper and Brush, 1991; Zimmerman and Canuel, 2000). Geochronologies from the hypoxic area on the continental shelf of the northern Gulf of Mexico also indicate that the current seasonal hypoxia, which can cover over 20,000 km^2, did not form annually prior to the 1950s (Sen Gupta et al., 1996), and models indicate not earlier than the 1970s (Justić et al., 2001, Scavia et al., 2003; Turner et al., 2006). Hypoxia was recorded with the first oxygen measurement made in the area in the summer of 1973 on the central Louisiana continental shelf (Rabalais et al., 2002) and remains an annual event.

Many systems that are currently hypoxic were not so when first studied. For systems with historical data from the first half of the twentieth century, declines in oxygen concentrations started in the 1950s and 1960s for the northern Adriatic Sea (Justić et al., 1987), between the 1940s and 1960s for the northwest continental shelf of the Black Sea, and in the 1970s for the Kattegat (Baden et al., 1990). Declining dissolved oxygen levels were noted in the Baltic Sea as early as the 1930s (Fonselius, 1969), but it was in the 1950s that hypoxia became widespread (Karlson et al., 2002). Other systems had hypoxia since the beginning of oxygen measurements, for example, in the 1900s for Kamak Bay, Korea (Lim et al., 2006); 1910s for Oslofjord, Norway (Mirza and Gray, 1981); 1920s for Thames Estuary, England (Andrews and Rickard, 1980); 1930s for Chesapeake Bay (Newcombe and Horne, 1938); and 1970s for the northern Gulf of Mexico (Turner et al., 2008).

The negative effects of hypoxia include loss of suitable and required habitat for many bottom-dwelling fishes and benthic fauna, habitat compression for pelagic fishes, direct mortality, increased predation, decreased food resources, altered trophic energy transfer, altered bioenergetics (physiological, development, growth, and reproductive abnormalities), and altered migration. These result in stressed fisheries species (Jørgensen, 1980; Caddy, 1993; Rabalais and Turner, 2001; Cheng et al., 2002; Kodama et al., 2002; Breitburg et al., 2009). Increasing nutrient loads that also change the nutrient ratios can affect the composition of the phytoplankton community and can shift trophic interactions (Turner et al., 1998). Hypoxia also alters or interrupts ecosystem functions and services such as nutrient cycling and bioturbation (Gutiérrez et al., 2000; Rabalais, 2004; Nizzoli et al., 2007; Middelburg and Levin, 2009; Weissberger et al., 2009; Sturdivant et al., 2012). Much of the alteration in functions from hypoxia can be observed in time-lapse videos captured with Wormcam by Sturdivant et al. (2012).

The frequency and duration of hypoxic events vary among systems, over time, and with varying nutrient loads or organic accumulation. Hypoxia ranges from aperiodic events with years to decades between reoccurrences to a persistent year-round feature that can last for years or centuries at a time. Dominant faunal responses differ by type of hypoxia (Figure 4). Aperiodic hypoxia, resulting from unusual or uncommon climate conditions, elicits the most dramatic response of mass mortality in sessile and, at times, mobile species. For benthic invertebrates, this dramatic response is due to the large numbers of sensitive species usually present prior to the hypoxic event. For example, the onetime hypoxic event in the New York Bight in 1976 that covered about 1,000 km^2 caused mass mortality of many commercial and noncommercial species (Boesch and Rabalais, 1991).

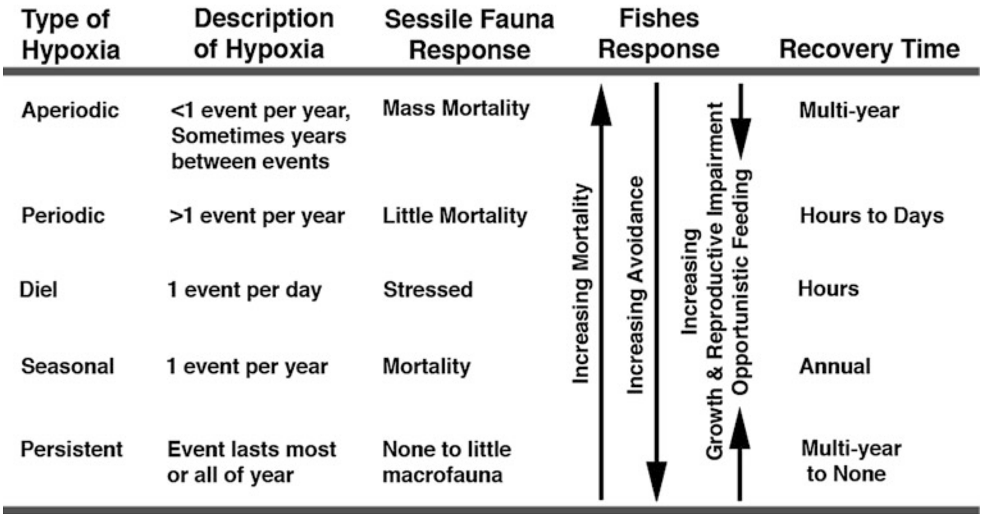
Summary

Hypoxia occurs in a wide range of systems and varies in temporal frequency, seasonality, and persistence. In temperate latitudes, bottom waters can remain hypoxic or anoxic for hours to months during summer and autumn.

There is no doubt that the increase in the areas within coastal and open oceans with hypoxia is real. Coastal water quality with regard to oxygen is currently on the decline, and the future, based on the continued increase in the global occurrence of hypoxia and current and projected increased loads of nutrients, is trending to more hypoxia. The formation of hypoxic areas has been exacerbated by any combination of interactions that increase primary production and accumulation of organic carbon leading to increased respiratory demand for oxygen below a seasonal or permanent pycnocline.

The overall forecast is for hypoxia to worsen, with increased occurrence, frequency, intensity, and duration. The consequences of global warming and climate change are effectively uncontrollable at least in the near term. On the other hand, the consequences of eutrophication-induced hypoxia can be reversed if long-term, broad-scale, and persistent efforts to reduce substantial nutrient loads are developed and implemented. The need for water and resource managers to reduce nutrient loads even if at a minimum, to maintain the current status, is critical in view of globally expanding hypoxia.

Anoxia, Hypoxia, And Dead Zones, Figure 3 Global distribution of documented cases of hypoxia related to human activities, *red dots*. Systems that have recovered from hypoxia through management of nutrient and organic loadings are *blue dots*. Systems that are eutrophic and in danger of becoming hypoxic are *yellow dots*. The number of hypoxic areas is cumulative for the successive time periods (From Diaz et al., 2010).



Type of Hypoxia	Description of Hypoxia	Sessile Fauna Response	Recovery Time
Aperiodic	<1 event per year, Sometimes years between events	Mass Mortality	Multi-year
Periodic	>1 event per year	Little Mortality	Hours to Days
Diel	1 event per day	Stressed	Hours
Seasonal	1 event per year	Mortality	Annual
Persistent	Event lasts most or all of year	None to little macrofauna	Multi-year to None

Anoxia, Hypoxia, And Dead Zones, Figure 4 Types of hypoxia and generalized faunal response. Sessile fauna are primarily macrobenthos. *Arrows* indicate direction of increased impact on fishes. Mortality in fishes is more likely from aperiodic hypoxia, with complete avoidance of persistent hypoxia. Physiological impairment and opportunistic feeding are greatest for periodic and diel hypoxia (Modified from Diaz and Breitburg, 2009).

The future pervasiveness of hypoxia in all ecosystems will depend upon a combination of climate change and land management. Climate change will affect water column stratification, organic matter production, nutrient discharges, and rates of oxygen consumption. Land management will also affect the concentrations of nutrients through agriculture.

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Cross-references

Climate Change
Dissolved Oxygen
Eutrophication
Nitrogen
Nutrient Dynamics
Nutrients
Oxygen Depletion
Phosphorus

ANTHROPOGENIC IMPACTS

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Definition

Anthropogenic impacts in this volume refer to the adverse effects of human activities on estuarine environments.

Introduction

Estuaries rank among the most heavily impacted aquatic ecosystems on earth, being affected by a wide array of anthropogenic activities both in adjoining coastal watersheds and in the water bodies themselves (Kennish, 2002). Most of these activities can be linked to ongoing rapid human population growth and development of the coastal zone. For example, about four billion people live within 60 km of the world's coastlines (Kennish, 2002; Kennish et al., 2008). In the USA, more than 125 million people now reside in coastal counties nationwide. People inhabiting low-lying coastal areas are more vulnerable to sea-level rise, coastal storms and storm surges, inundation and flooding, and other coastal hazards.

While many people simply prefer to live near the ocean, others inhabit the coastal zone for economic opportunity involving tourism, recreational and commercial fisheries, mariculture, transportation and shipping, domestic and industrial construction, electric power generation, oil and gas recovery, and other human activities. Escalating human settlement has altered land use/land cover in coastal watersheds, creating impervious surfaces that facilitate nonpoint-source pollution input to estuaries. Much of this is coupled to developing infrastructure such as construction of roadways and bridges, electric utilities, as well as water, sewer, and gas lines. Other major pollutant delivery systems are point-source inputs. Some human activities have physically altered estuarine shorelines and habitats, impacting biotic communities. Included here are the construction of bayshore housing, lagoons, bulkheads and other shore protection features, harbor and marina development, dredging and dredged material disposal, wetland reclamation, and channel and inlet stabilization. Upland modifications (e.g., dams and reservoirs, deforestation and habitat fragmentation, and channelization) can significantly exacerbate other anthropogenic stresses leading to greater pressures in estuarine systems. Increasing freshwater diversions for agriculture and other human needs, such as in California (USA), can significantly alter salinity, circulation, and biotic communities in estuaries.

Estuarine impacts

Kennish et al. (2014a) identified 12 major anthropogenic stressors on estuarine ecosystems. These include (1) eutrophication; (2) sewage and organic wastes; (3) habitat loss and alteration, shoreline hardening, and erosion; (4) chemical contaminants; (5) human-induced sediment/particulate inputs; (6) overfishing; (7) intensive aquaculture; (8) introduced/invasive species; (9) human-altered hydrological regimes; (10) climate change; (11) coastal subsidence; and (12) floatables/debris. An additional stressor of importance is a group of pathogens that can impact human use of estuaries. According to Kennish and Paerl (2010), anthropogenic stressors can be categorized into those that degrade water quality and are primarily chemical and biological in nature (e.g., nutrient enrichment, chemical contaminants, and pathogens), impact habitat and are mainly physical factors (e.g., shoreline hardening, lagoon construction, dredging and dredged-material disposal), and alter biotic communities and are effectively driven by multiple stressors (e.g., overfishing, introduced/invasive species, human-altered hydrological regimes, and climate change).

Eutrophication

Nutrient enrichment, notably nitrogen and phosphorus, has led the eutrophication of many estuaries, particularly shallow systems with long water residence times (Kennish and de Jonge, 2011). Reactive nitrogen inputs to estuaries have been increasing through time, resulting in escalating eutrophication of estuarine water bodies (Howarth et al., 2002a; Rabalais, 2002; Bricker et al., 2007; Burkholder et al., 2007; Howarth, 2008; Anderson et al., 2010). Of the 138 estuaries in the USA examined by Bricker et al. (2007), 84 were determined to be highly eutrophic. Greater nitrogen and phosphorus loads to estuaries and coastal marine waters are attributed to accelerating coastal development and intensification of agriculture (Howarth et al., 2002b; Galloway et al., 2008; Howarth, 2008; Anderson et al., 2010). Aside from agriculture sources, these nutrients derive from lawn fertilization, stormwater runoff, municipal and industrial wastewaters, malfunctioning septic systems, groundwater seepage, and atmospheric deposition. The eutrophication of estuaries is manifested by an array of damaging cascading changes in ecosystem structure and function such as decreased dissolved oxygen levels, increased microalgal and macroalgal abundance, occurrence of harmful algal blooms (HABs), loss of seagrass habitat, reduced-biodiversity, declining fisheries, imbalanced food webs, altered biogeochemical cycling, and diminished ecosystem services (Kennish and de Jonge, 2011).

Sewage and organic wastes

Sewage and organic wastes also increase nutrient and organic carbon loading that can exacerbate estuarine eutrophic conditions (Kennish, 2001a). Organic carbon enrichment is coupled to elevated biochemical oxygen demand and depleted dissolved oxygen levels in some water bodies. These wastes derive from domestic and industrial sources, livestock and fish processing facilities, wildlife inputs, aquaculture operations, and other sources. Hypoxia of estuarine and coastal marine environments has increased worldwide over the past several decades due to eutrophication driven by nutrient enrichment and organic carbon loading (Diaz and Rosenberg, 1995, 2008).

The calefaction of estuarine waters by the release of heated effluent from electric generating stations and other industrial facilities can also deplete dissolved oxygen levels. Thermal discharges from electric generating stations have commonly increased mortality of susceptible estuarine organisms in near-field regions of outfall sites. Aside from increased mortality due to reduced dissolved oxygen, heat-shock and cold-shock mortality at electric generating stations has occasionally caused mass mortality of finfish populations which cannot adapt to the rapid changes in water temperature associated with abrupt changes in station operation (Kennish, 1992).

Habitat loss and alteration

Coastal population growth and development have led to substantial estuarine habitat loss and alteration, impacting biotic communities. Physical alteration is associated with the dredging of channels and other subtidal areas, the construction of hardened shorelines, and the removal of vegetation and soils during construction of buildings, roadways, and other infrastructure elements in watersheds that convert natural habitats to impervious cover and increase erosion, runoff, and nonpoint pollution to estuarine water bodies. Water quality is often degraded as a result. Other adverse effects include habitat fragmentation, ecosystem isolation, and functional degradation of upland and wetland complexes (Kennish, 2001a). Some of these changes can be intractable.

Historically, salt marshes and other wetland habitat bordering estuarine basins have been altered extensively by grid ditching, marsh diking, draining and filling (for agriculture), impoundments for wildlife, and reclamation for domestic and industrial development. The hydrology of wetland systems has been invariably changed by construction of impoundment dikes, water-control embankments, levees, dams for flood control, as well as canals and spoil banks. Tidal flooding, water flow, and drainage are often modified, reducing sediment loading to the marsh surface and even arresting vertical accretion and hastening marsh submergence (Kennish, 2001b). Concomitantly, the loss of marsh in many regions is accelerating due to climate change-driven sea-level rise. Human activities have eliminated more than 50 % of the original tidal marsh habitat in the USA and more than 70 % of the mangroves fringing the coast of Puerto Rico (Kennish, 1997; Alongi, 1998). The destruction of wetland habitat decreased substantially in the USA after enactment of the 1972 Clean Water Act.

Localized areas along estuarine shorelines have been affected by construction and use of docks, piers, boat ramps, and marinas. Other shorelines have been modified by marine engineering structures such as bulkheads and revetments for bank stabilization, jetties at inlets, stormwalls, and other protective features. Constructing physically static structures in dynamic estuarine environments has impacted the function of sensitive habitats often diminishing their ecological value.

Chemical contaminants

Many estuaries receive a wide range of chemical contaminants because they are located in close proximity to heavily populated metropolitan centers and other developed coastal areas. Urbanized estuaries are often the most heavily impacted. Major sources and delivery systems include agricultural and urban runoff, municipal and industrial discharges, groundwater inputs, riverine inflow, and atmospheric deposition. Chemical contaminants entering estuaries may concentrate in the water column, accumulate in bottom sediments and organisms, or exit to coastal waters. Bottom sediments of estuaries are typically repositories of chemical contaminants because many of these substances are particle reactive, sorbing to grain surfaces, and ultimately settling to the estuarine floor.

Among the most important groups of chemical contaminants found in estuarine environments are halogenated hydrocarbons, polycyclic aromatic hydrocarbons (PAHs), and metals. These contaminants are potentially damaging to estuarine habitats and hazardous to estuarine and marine organisms. Substances such as aromatic and aliphatic hydrocarbons derived from oil spills and seepages, as well as volatile organic compounds, can be acutely toxic to biotic communities (Kennish, 1997).

The halogenated hydrocarbons are a ubiquitous group of environmental contaminants consisting of low- to high-molecular-weight compounds. Examples are organochlorine biocides (insecticides, herbicides, and fungicides), low-molecular-weight compounds (chlorofluorocarbons), and high-molecular-weight chemicals (chlorinated aromatics and chlorinated paraffins). PCBs and DDT are two of the most notable halogenated hydrocarbons that have historically impacted estuarine environments (Kennish, 2001a).

PAHs consist of a group of chemical carcinogens, mutagens, and teratogens that originate from both natural (e.g., volcanic activity and oil seeps) and anthropogenic sources, although the inputs from anthropogenic sources (e.g., fossil fuel combustion, waste incineration, municipal and industrial wastewaters, and land runoff) typically predominate in estuaries (Kennish, 1992). The low-molecular-weight PAHs are more toxic than the high-molecular-weight forms. Hepatic neoplasia and other diseases in aquatic organisms have been attributed to PAH exposure (Kennish, 1997). A range of biochemical, physiological, behavioral, and other sublethal responses has also been documented in estuarine organisms that can adversely affect biotic community structure.

Oil spills and leakages from marine vessels and fixed installations, as well as from nonpoint-source inputs from coastal watersheds, are hazardous to estuarine organisms and habitats. The lethal and sublethal effects of polluting oil on estuarine and marine organisms are well established (Kennish, 1992, 1997, 2001a). Both aromatic and aliphatic components are problematic as noted above. Benthic organisms are particularly susceptible to oil accumulation, and contaminated habitats such as salt marsh systems can be adversely impacted by the oil for decades due to oil-contaminated sediments which are hazardous to settlement and recruitment of the organisms.

Metals

The literature is replete with pathological responses of estuarine and marine organisms to toxic levels of metals, including neurological, digestive, reproductive, and respiratory disorders, tissue inflammation and degeneration, and developmental abnormalities. Feeding behavior and growth inhibition are commonly observed. Transition metals (e.g., copper, cobalt, iron, and manganese), metalloids (e.g., arsenic, cadmium, lead, mercury, selenium, and tin), and organometals (e.g., methylmercury, tributyltin, and alkylated lead) can be toxic, particularly the organometals (Kennish, 1997; Kennish, 1998; Kennish et al., 2008). Metals are persistent in estuarine environments. They tend to bioaccumulate in organisms, and some metals such as methylmercury undergo biomagnification, with highest levels found in upper-trophic-level organisms that often serve as a food source for humans. Hence, the health of humans consuming metal-contaminated seafood can be impaired. There are many potential sources of metals in estuaries, including industrial activity (e.g., mining, smelting, refining, and electroplating operations), fossil fuel combustion, landfill leachates, shipping, marinas, and ash disposal. Delivery pathways are river discharges, groundwater inputs, and atmospheric deposition.

Human-induced sediment/particulate inputs

Human activities in coastal watersheds have facilitated inputs of sediments and other particulates to estuaries. The removal of natural vegetation and increase in impervious cover with watershed development hasten the delivery of sediments to estuarine basins. Silviculture operations, particularly in developing countries, have dramatically increased sediment loads to coastal areas (Kennish et al., 2008). One of the adverse effects is altered water and sediment quality. An increase in water column turbidity leads to the attenuation of light and shading of the estuarine floor that can reduce primary production and cause a decline of seagrasses and other essential benthic habitat. For example, Moore et al. (2012, 2014) correlated dieback of seagrasses in the Chesapeake Bay system in part to elevated turbidity levels.

Overfishing

Overfishing or overharvesting of finfish and shellfish populations not only results in depleted stocks but also the alteration of the food web structure of estuaries. While estuaries historically have had exceptional recreational and commercial fisheries, overexploitation of the biotic resources has been a concern through time. In mid-Atlantic coastal bays, overharvesting of shellfish, together with disease and predation, has been implicated in the dramatic decline of oyster and hard clam populations. Overfishing may have played an important role in the decline of Chinook salmon (*Oncorhynchus tshawytscha*), delta smelt (*Hypomesus transpacificus*), and striped bass (*Morone saxatilis*) fisheries in San Francisco Bay (USA) after the 1970s. Similarly, overfishing may have been a factor in the reduction of commercial finfisheries in Albemarle-Pamlico Sounds, North Carolina, and Sarasota Bay, Florida (Kennish, 2000).

Intensive aquaculture

A significant amount of the seafood consumed by humans (>25 %) now derives from aquaculture, which may partially compensate for overfishing (Engelman et al., 2008). Shellfish aquaculture predominates in countries of the Far East (e.g., China, Vietnam, and the Philippines). Much finfish aquaculture also occurs in countries of the Far East, but in many other countries as well. Intensive aquaculture has caused considerable coastal pollution, water and sediment quality degradation, and diseases resulting from the feces and uneaten food of the feedlot operations (New, 2002).

Aside from degraded water and sediment quality, aquaculture operations also cause habitat conversion and changes in hydrological regimes.

Introduced/invasive species

Organisms that are not endemic to an estuary, but are introduced or invade the water body, can have significant ecological impacts. Many species are introduced for commercial or recreational interests, an example being the introduction of the striped bass (*Morone saxatilis*) to San Francisco Bay (USA). In fact, the dominant species of organisms in San Francisco Bay are mainly introduced forms, with more than 200 nonindigenous species now inhabiting bay waters and wetland habitat (Kennish, 2000). Some exotic species inhabiting estuaries have been accidentally introduced via ballast water or some other means. Nearly all estuaries are affected by introduced or invasive species (Carlton and Geller, 1993; Kennish, 2002; Kennish et al., 2008)

Introduced and invasive species can be a danger to the stability and biodiversity of an estuarine ecosystem. In those cases where native controls are lacking, these species can have a significant competitive advantage, often rapidly dominating plant or animal communities. The food web structure is commonly disrupted, and native species may be displaced or greatly reduced in abundance. Changes in species composition and distribution commonly occur (Cohen and Carlton, 1998). Species diversity in these systems may drastically decline as well via intense competition and predation. For instance, the Asian clam (*Potamocorbula amurensis*), introduced into Suisun Bay (USA), has decimated the phytoplankton community and outcompeted the native shellfish species (*Macoma balthica* and *Mya arenaria*).

The introduction or invasion of exotic species is expected to increase in the future due to an expanding world population, the effects of climate change, and greater shipping and other human activities at sea and in estuaries. These changes will likely promote additional ecological disruption. More mariculture ventures, particularly in developing countries, will also add to these effects (Kennish, 2002).

Human-altered hydrological regimes

As population growth increases in coastal regions, so does the demand for freshwater to meet domestic, industrial, and agricultural needs. The urbanization of coastal watersheds also results in greater impervious land cover leading to accelerated freshwater runoff and higher river discharges. The increase in freshwater flow decreases the water residence time in estuaries, while increasing their capacity to dilute, transform, or flush contaminants (Kennish, 2000). In contrast, dams and reservoirs constructed upland reduce downstream freshwater flow. Other changes that can significantly alter water-flow regimes along coasts include channelization, marsh impoundment, and wetland habitat destruction which affect natural water storage capacity (Kennish et al., 2008). Shifts in water quality and quantity are important drivers of change in the abundance, distribution, reproductive success, and productivity of estuarine organisms (Kennish, 2000). Estuarine circulation can also significantly change. The impact of freshwater diversions is perhaps best exemplified by San Francisco Bay (USA), where 50 % of the freshwater flow has been diverted for human use, resulting in a modified salinity regime as well as altered biotic communities in the bay.

Freshwater diversions are used for ecological applications as well, such as coastal wetland restoration. Teal et al. (2012), for example, discuss diversions of river water into coastal wetlands as part of plans to mollify the areal loss of coastal wetlands in Louisiana by reversing or slowing the rates of degradation. However, freshwater diversions such as those noted by Teal et al. (2012) can cause major changes in estuarine water quality, notably large reductions in salinity and increased nutrient availability that affect plant communities, herbivory, and overall marsh stability.

The use of flood control structures has been shown to completely modify some estuarine ecosystems. For example, acute changes in water inflow to the Haringvliet, Grevelingenmeer, and Oosterschelde estuaries in southwest Netherlands occurred subsequent to the construction of dikes to avert flooding problems, such as those incurred in 1953, resulting in major changes in the physical-chemical conditions and biotic communities of the water bodies. While the Oosterschelde remained a productive estuary after these structural changes, the Haringvliet basin was altered to a highly polluted freshwater body, and the Grevelingenmeer became a saltwater system. Tidal marshes and tidal flat habitats also changed considerably (Kennish et al., 2008). Changes such as those observed in the Haringvliet, Grevelingenmeer, and Oosterschelde estuaries demonstrate the magnitude of human influence on coastal environments, particularly in regard to human-altered hydrologic regimes.

Climate change

An accumulating database indicates that human factors are important drivers of change of world climate (Skinner, 2012). Increasing global temperatures, ascribed in large part to carbon dioxide emissions, have been linked to greater frequency and severity of damaging storms, coastal flooding, droughts and fires, and other hazards projected by climate forecasting models for the twenty-first century (IPCC, 2007). Extreme climate events and ongoing sea-level rise will be hazardous to coastal communities worldwide.

During the twentieth century, global sea-level rise amounted to 0.5–30 cm, being largely attributed to the increase in global surface temperatures (mean = +0.6 ± 0.2 °C), melting of glaciers and ice sheets, and thermal expansion of the oceans (IPCC, 2007). Global sea-level rise during the twenty-first century is projected to increase by 52–98 cm (IPCC, 2013). Relative sea-level rise will be even greater in some regions due to coastal subsidence, as in the case of New Jersey and other mid-Atlantic states in the USA.

Rising sea level and coastal inundation will lead to significant loss of some coastal wetlands, eliminating buffer and rendering coastal communities more vulnerable to extreme events. Human-induced climate change will also alter temperature and salinity regimes and the structure and function of biotic communities in estuaries (Kennish, 2002). Configurations of estuarine basins will be modified as they widen and deepen. Shifts will occur in nutrient and sediment supply as well as freshwater inputs. Tidal prisms and tidal ranges will change in many systems. More frequent flooding and inundation of bayshore areas will pose hazards to vulnerable coastal communities worldwide (Kennish et al., 2008).

Coastal subsidence

Subsiding coasts result in similar impacts as rising sea level on estuarine and wetland systems. For example, as coastal subsidence increases, estuarine shoreline retreats and land submergence accelerates causing the loss of fringing wetland habitat. The wet surface area of the estuary expands, together with the basin volume, thereby altering the system bathymetry and configuration. The salinity regime, circulation, and other physical-chemical characteristics of the estuary can change considerably as well, which will also affect biotic communities.

The effects of coastal subsidence are becoming more evident around the world with accelerating population growth and development in coastal watersheds. Human activities have contributed greatly to the subsidence problems in some areas via excessive groundwater withdrawal for domestic and agricultural use and oil and gas extraction. Natural factors, such as subsurface sediment compaction, crustal (tectonic) movements, and sinkhole formation by karst processes, are more significant in some regions. In the USA, subsidence has been well chronicled at Galveston Bay, Texas, due to oil and gas extraction (Shipley and Kiesling, 1994), along the Louisiana coast due to sediment compaction (DeLaune and Pezeshki, 1994), and at Chesapeake Bay due to groundwater withdrawal and other factors (Boon et al., 2010).

Significant coastal subsidence has also been documented in other countries (e.g., Po Delta, Italy, and Tokyo, Japan) (Kennish et al., 2008). Coastal subsidence problems will take on added significance during the twenty-first century, exacerbating those due to eustatic sea-level rise (Kennish, 2002). Subsidence rates, such as those noted by Boon et al. (2010) at Chesapeake Bay (−1.3 to −4.0 mm year), will likely continue unabated into the future. It will be necessary for coastal decision makers to carefully consider the management options necessary to reduce their impacts.

Floatables/debris

Marine debris, notably plastics, has become an international problem in estuarine and marine environments. The use of plastic products has reached an all-time high in many developed countries; plastic debris has been an ongoing problem in coastal waters of the USA (Ribic, 1998; Kennish, 2001a). Plastics are particularly damaging to aquatic environments because they essentially do not degrade, thereby polluting habitats for many years. They pose a threat to many organisms, particularly fish, turtles, birds, and mammals that ingest some of the materials or become entangled in fishing line, nets, and packing bands. Some organisms mistake floatables for prey (Shaw and Day, 1994). The ingestion of plastics and other marine debris can suffocate the animals or obstruct their digestive systems, causing death.

Pathogens

Estuaries worldwide are susceptible to the entry of pathogenic microorganisms (i.e., bacteria and viruses) from land runoff and sewage wastes that pose a risk to human health. Fecal pollution (i.e., fecal coliform bacteria, enterococci, and coliphage) and human enteric pathogens and enteroviruses can greatly impair human use of impacted estuarine and coastal marine waters (Lipp et al., 2001). Faulty septic systems, sewage treatment plant effluent, and wildlife wastes often degrade estuarine water quality, which must be continually monitored. Treated municipal wastewater and urban stormwater runoff may contain more than 100 enteric pathogens (National Research Council, 1993). These organisms, which are widely distributed by coastal storms (Fries et al., 2007), pose a hazard to humans who swim in contaminated estuarine waters or consume contaminated seafood products.

Parasites, notably helminths and protozoa, are commonly associated with waterborne diseases as well. Helminths linked to untreated sewage in estuarine and other aquatic systems include hookworms, roundworms, tapeworms, and whipworms. Pathogenic enteric protozoa derived from sewage contamination can be equally devastating to human health.

Summary

A wide array of anthropogenic factors contributes to estuarine degradation. Chief among these are (1) nutrient loading and eutrophication; (2) sewage and organic wastes; (3) habitat loss and alteration, shoreline hardening, and erosion; (4) chemical contaminants; (5) human-induced sediment/particulate inputs; (6) overfishing; (7) intensive aquaculture; (8) introduced/invasive species; (9) human-altered hydrological regimes; (10) climate change; (11) coastal subsidence; and (12) floatables/debris. Pathogens coupled to human waste discharges and wildlife are also significant since they adversely affect water quality. Anthropogenic stressors can be categorized into those that degrade water quality and are primarily chemical and biological in nature (e.g., nutrient enrichment, chemical contaminants, and pathogens), impact habitat and are mainly physical factors (e.g., shoreline hardening, lagoon construction, dredging and dredged-material disposal), and alter biotic communities and are effectively driven by multiple stressors (e.g., overfishing, introduced/invasive species, human-altered hydrological regimes, and climate change). Human activities can significantly impact the structure, function, and ecological health of estuaries.

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Cross-references

Anoxia, Hypoxia, and Dead Zones
Biochemical Oxygen Demand
Bulkheads
Clean Water Act
Climate Change
Dredging
Eutrophication
Extreme Events (Hurricanes)
Invasive Species
Marsh Sediment Toxicity
Nonpoint Source Pollution
Oil Pollution
Pathogens
Polycyclic Aromatic Hydrocarbons
Revetments
Storm Surges
Toxic Blooms
Trace Metals in Estuaries
Water Quality
Wetlands Reclamation

ARCHAEA

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Synonym

Archaebacteria

Definition

Archaea constitute one of the three domains of life, as phylogenetically distinct from the domains Bacteria and Eukarya (Woese and Fox, 1977; Woese et al., 1990).

Etymology

Archaea (singular: archaeon) are derived from Greek ἀρχαῖος – ancient, primitive.

The term was coined to reflect the apparent antiquity of traits in archaeal species and their aptness to inhabit environments like those presumed for early Earth.

Phylogeny

On the phylogenetic tree of organisms, Archaea form one of the three fundamental branches called domains (Woese et al., 1990). Further branching within the archaeal domain is largely based on ribosomal RNA sequences, but also relies on evidence from comparative analysis of protein sequences. Typical markers in archaeal phylogeny are ribosomal proteins, histones, and components of the transcription system and the cell division apparatus (Brochier-Armanet et al., 2011). In addition to the initially recognized phyla Crenarchaeota and Euryarchaeota (Woese et al., 1990), four candidate phyla have been proposed: Kor-, Nano-, Thaum-, and Aigarchaeota (Brochier-Armanet et al., 2011). Given the small number of 187 sequenced archaeal genomes (Genomes OnLine Database; May 2013), archaeal phylogeny continues to be uncertain.

Physiology and metabolism

Archaea are single-celled organisms showing prokaryotic structure, but differ from bacterial cells by the absence of peptidoglycan from their cell wall and by the presence of histones associated with their DNA. In contrast to the phospholipid bilayer confining bacterial and eukaryotic cells, archaeal cell membranes consist of a single layer of isoprenoid diethers and tetraethers. Most archaeal genomes contain clustered, regularly interspaced palindromic repeats (CRISPR) that confer adaptive immunity against virus infections (Sorek et al., 2008; Marraffini and Sontheimer, 2010).

No chlorophyll-like pigments or photosynthetic electron transport chains are known in Archaea. However, some archaea possess bacteriorhodopsin, which functions as light-driven proton pump across the cell membrane. The reflux of protons can be exploited for ATP synthesis, completing the conversion of light energy into chemical energy for cellular metabolism.

Archaea show diverse forms of chemotrophic metabolism, including both the formation and the anaerobic oxidation of methane, both oxidation and reduction of CO, the aerobic oxidation of H_2 (knallgas reaction), most of the oxidative and reductive processes in the N and S cycle, and redox reactions of various metals and metalloids (Amend and Shock, 2001; Stolz et al., 2006; Oelgeschläger and Rother, 2008; Conrad, 2009; Knittel and Boetius, 2009; Bini, 2010).

In sulfur-based energy metabolism, oxidized and intermediate S compounds (sulfate, sulfite, thiosulfate, and elemental sulfur) are reduced by H_2 or organic compounds, while reduced and intermediate S compounds (sulfide, elemental sulfur, and thiosulfate) can be oxidized by O_2, nitrate, or ferric iron (Amend and Shock, 2001). Oxidation and reduction of S compounds are widely used energy sources for archaea in hydrothermal and marine ecosystems.

Several processes of nitrogen-based energy metabolism are known in archaea inhabiting extreme and moderate environments, including aerobic ammonia oxidation, dissimilatory reduction of nitrate to nitrite and of nitrite to ammonium, and all steps of denitrification. Archaea may also use nitrate assimilation or N_2 fixation to meet their cellular N demand (Zumft, 1997; Amend and Shock, 2001; Philippot, 2002; Cabello et al., 2004; Francis et al., 2005). One handful of cultured and numerous uncultured members of Thaumarchaeota share the metabolic capacity of aerobic ammonia oxidation. Given their ubiquity and abundance in diverse habitats, archaeal ammonia oxidizers may play a major role in the global N cycle (Francis et al., 2005; Prosser and Nicol, 2008; Stahl and de la Torre, 2012).

Methanogenesis is catalyzed exclusively by Archaea and may proceed via three different pathways: reduction of CO or CO_2, reduction of methyl groups in small organic compounds, or cleavage of acetate. The strictly anaerobic methanogenic archaea occur abundantly in natural and agricultural wetlands, hydrothermal environments, and the digestive tracts of ruminants and termites (Liu and Whitman, 2008; Conrad, 2009). The oxidation of methane with O_2 is known only in Bacteria, whereas its anaerobic oxidation with sulfate, iron, or manganese is catalyzed by diverse and mostly uncultured Archaea, with or without the help of bacterial symbionts (Beal et al., 2009; Knittel and Boetius, 2009; Milucka et al., 2012).

Ecology

Archaea occur in virtually any habitat that supports life. They are adapted to salt concentrations of up to 5.2 M, to temperatures above 120 °C, and to life in permafrost environments. Archaeal species inhabit mine drainage of pH <0 and soda lakes of pH >11 and thrive at hydrostatic pressures of up to 120 MPa in the deep sea and below the sea floor (Valentine, 2007; Lipp et al., 2008; Bowers and Wiegel, 2011; Takai and Nakamura, 2011). Although best known for their endurance of one or more extreme conditions, Archaea are also ubiquitous and vital parts of communities in moderate environments. Their metabolic activities contribute to the major element (C, N, S, Fe) cycles on local to global scales.

Representatives of Cren-, Eury-, Nano-, and Thaumarchaeota engage in mutualistic and parasitic symbioses with ruminants, insects, corals, sponges, molluscs, ciliates, sulfate-reducing and sulfide-oxidizing bacteria, and among archaea (Moissl-Eichinger and Huber, 2011; Wrede et al., 2012). Archaea are also hosts of diverse viruses (Pina et al., 2011) and subject to predation by zooplankton and zoobenthos. To date, neither predatory nor pathogenic Archaea are known.

Current knowledge on Archaea in moderate marine, estuarine, and freshwater environments is largely confined to methanogenic Euryarchaeota and ammonia-oxidizing Thaumarchaeota. Methanogens grow in the strict absence of O_2 and rely on H_2 or acetate produced by fermenting organisms. Their effective competition for these substrates decreases with increasing availability of electron acceptors other than CO_2. Between the three metabolic types of methanogenic archaea, relative abundance and activity appear to depend on redox potential, pH, temperature, salinity, and the concentrations of phosphate, sulfate, and organic C compounds (Liu and Whitman, 2008; Torres-Alvarado et al., 2013).

The ubiquity, abundance, and diversity of ammonia-oxidizing archaea in aquatic environments have largely been inferred from abundance and diversity of archaeal *amoA*, a gene encoding the key enzyme of aerobic ammonia oxidation. Although the postulated activity was confirmed in all four cultured species, the biochemistry of this enzyme allows for the turnover of several other substrates, including methane. The archaeal enzyme shows much higher substrate affinity than its bacterial counterpart, making archaea strong competitors at low concentrations of ammonium or O_2. Distribution patterns of archaeal and bacterial *amoA* suggest greater tolerance to sulfide, acidity, and high temperatures in archaeal compared to bacterial ammonia oxidizers (Francis et al., 2005; Erguder et al., 2009; Martens-Habbena et al., 2009; Pester et al., 2011; Hatzenpichler, 2012; Stahl and de la Torre, 2012). In estuaries, steep gradients of multiple environmental factors coincide to confound evidence for possible effects of salt or sulfide (Bernhard and Bollmann, 2010).

Summary

Archaea constitute one of the three domains of life, as distinct from the domains Bacteria and Eukarya both phylogenetically and by cell structure. Still limited sequence information suggests six phyla within the archaeal domain.

Archaea show diverse forms of chemotrophic metabolism, including both the formation and the anaerobic oxidation of methane, oxidation and reduction of CO, aerobic oxidation of H_2, many oxidative and reductive processes in the N and S cycle, and redox reactions of various metals and metalloids. Bacteriorhodopsin facilitates a simple mechanism of phototrophic energy conversion without assimilation of CO_2.

While Archaea are most famous for enduring extreme temperature, salinity, pH, or pressure, they are also widespread in a broad range of moderate habitats. Their metabolism makes a relevant contribution to global biogeochemical cycles. Archaea interact with members of all domains in mutualistic or parasitic symbioses, host viruses, and fall prey to bacterivores, but are not known in the roles of predator or pathogen.

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Cross-references

Nitrate Reduction
Nitrification
Symbiosis

ARTIFICIAL REEF

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Synonyms

Artificial habitats; Artificial reef structures; Man-made habitats; Man-made reefs

Definition

"An artificial reef is one or more objects of natural or human origin deployed purposefully on the seafloor to influence physical, biological, or socioeconomic processes related to living marine resources" (Seaman and Jensen, 2000, 5). While this definition is directed toward the marine environment, a few modifications to include "estuarine and freshwater" resources make it applicable to the estuarine environment as well. Also, it should be noted that artificial reefs are not always "deployed purposefully" as "accidental" deployments of ships or other objects sunk by storms, acts of war, or other episodic events can also create submerged structure that function under the banner of artificial reefs. Additionally, a broader appreciation of the utility of artificial reefs would certainly make them useful to disciplines which are not mutually exclusive. Thus, artificial reefs can simultaneously influence physical, biological, *and/or* socioeconomic processes.

Introduction

Artificial reefs have likely been deployed for the purposes of enhancing fishing for millennia. Most probably, fish associated with naturally occurring debris in rivers and lakes were recognized early in human history as functioning as preferred structure for some fish species. It would have been intuitive to help Mother Nature along by deploying similar-looking, natural structures in aquatic habitats to further enhance fishing. The modern impetus for artificial reef seems to have come from the congregations of reef fishes observed around sunken ships and downed warplanes resulting from sea battles during the World War II in the Pacific Ocean. For example, Chuuk (Truk) Lagoon in Micronesia, where 32 merchant ships and 249 aircraft were sunk, has become a diver's "paradise" since the early 1970s (Trumbull, 1972). The advent and popular use of scuba by the general populace since the late 1950s helped the general recognition that artificial reefs were a "good thing" when it came to fisheries, but there were few data and studies directed toward truly establishing the verifiable reasons for their deployment. Of late, artificial reefs have garnered attention by natural resource managers, aquatic scientists, and the fishing public to improve fishing and fisheries around the world. The popular mantra is that "more reefs will mean more fish" without regard for the effects of these structures on other natural processes. Lastly, the "proof" of artificial reefs as an effective management tool is wanting, largely owing to the lack of scientifically valid opportunities to test various hypotheses.

Kinds of artificial reefs

Artificial reefs are generally of two basic types: benthic (i.e., located on the majority of mid-water/surface Fig. 1). The majority of mid-water and surface reefs serves as fish-aggregating device (FADs) and is directed primarily at the pelagic and epipelagic game fishes in coastal areas. Benthic reefs have been the subject of much effort in design and planning. A broad variety of structure have been used in artificial reefs, but generally they can be considered as either structure of opportunity (i.e., made of refuse materials) or designed and engineered specifically to serve as artificial reefs. Structure of opportunity include (but are not limited to) ships, automobiles, and other vehicles (e.g., railroad cars and airplanes), derelict oil and gas platforms, bridge rubble, remnant construction materials (e.g., broken sewer pipe), scrap metal of various composition and gauges, white goods (refuse household appliances such as washing machines, stoves, and refrigerators), vehicle tires, fiberglass materials, porcelain, and any materials considered dense enough to sink and withstand some degree of wave action or current surge.

Engineered structures can run the gamut of all of the above but are generally composed of structures thought to have a longer life span (durability) and greater stability (density) than reefs composed of structures of opportunity. Engineered structures are most often composed of concrete, concrete and rock aggregate, and heavy gauge steel.

Generally, artificial reefs are deployed as modules or units of a size and shape readily transportable and deployable. These modules are then organized into sets of two or more modules, and the sets are often organized into groups. Lastly, the groups of sets and modules comprise the entire reef complex that may be of considerable extent, covering several kilometers (Grove et al., 1991).

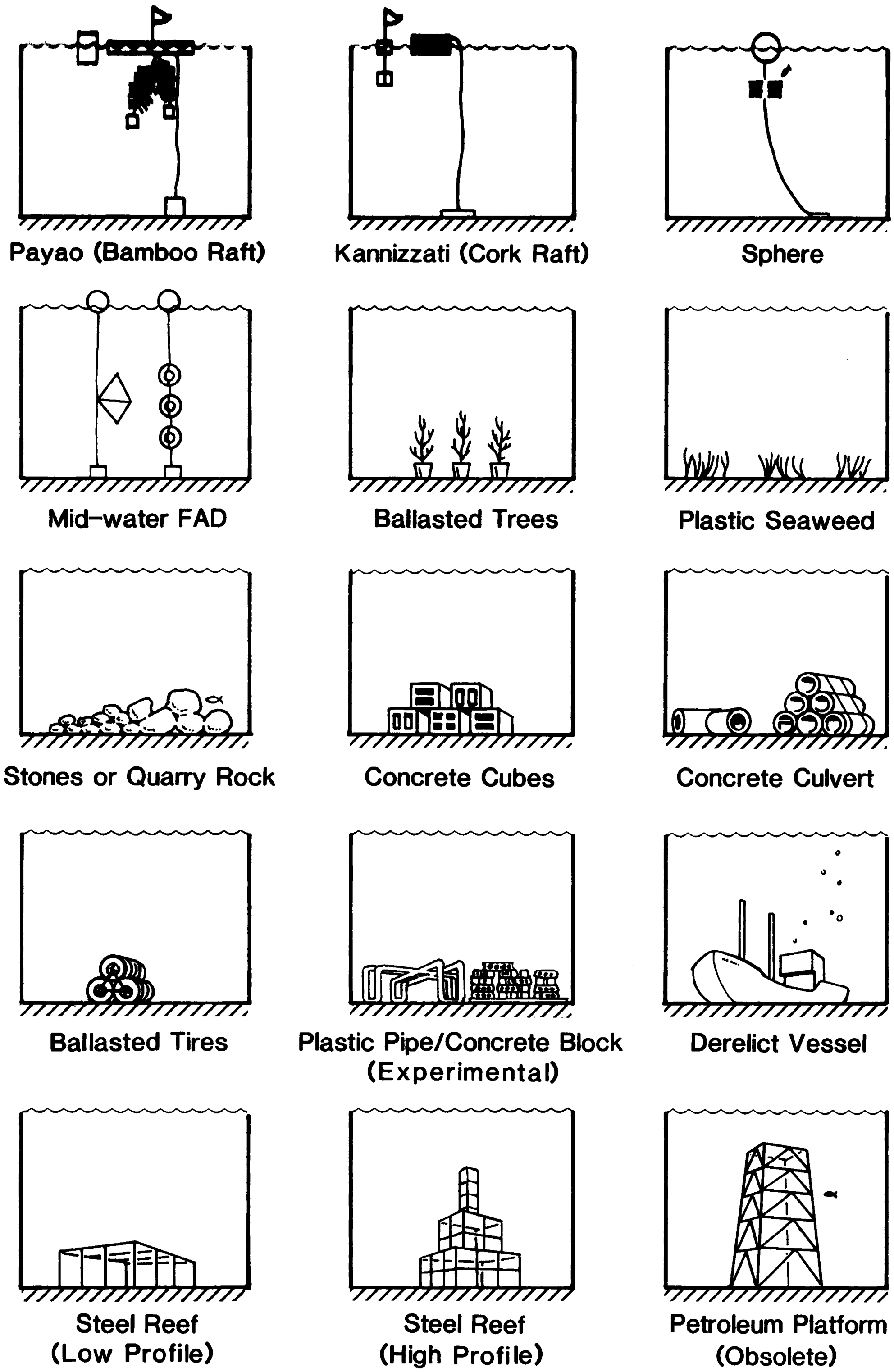
Locations of artificial reefs

Artificial reefs have been deployed in virtually every aquatic ecosystem from freshwater streams, rivers, ponds, and lakes to estuaries, fjords, bays, and the open ocean (both near coastal and far offshore). The substrate type upon which the reefs are deployed is an important consideration, as some substrates are easily eroded or scoured, and the reef can become quickly covered so as to become nonfunctional as an artificial reef.

Position of the reef in proximity to other biota is often a deployment consideration to facilitate colonization by juveniles, adults, or prey items. Depending on the intended function of the reef, positioning the reef to facilitate (or prohibit) access by users is often a consideration as well.

Functions of artificial reefs

Artificial reefs can have many functions via the ecosystem services they provide. Often a chief function is to enhance fisheries for fishing opportunities for both the commercial and recreational fishing public. Environmentally, artificial reefs can serve to mitigate damage to natural areas, serve to enhance biotic community diversity, or fulfill other goals of resource managers. Artificial reefs have been used as objects to deter various fishing activities such as the protection of seagrass beds from trawling (Fabi and Spagnolo, 2011). Recreationally, artificial reefs can serve as scuba and snorkeling sites, especially to enhance areas void of "interesting substrate" or as alternative dive sites to protect natural areas from potential damage by divers. Artificial reefs can provide a substrate to allow settling



Artificial Reef, Figure 1 Examples of various artificial reefs, including FADs or fish-aggregating device deployed chiefly to attract pelagic or epipelagic fish (From Seaman and Sprague, 1991).

of sessile (attached) organisms such as bivalve mollusks (Relini et al., 1994). Japanese researchers have investigated the deployment of artificial reefs as structures to help divert ocean currents to facilitate upwelling that brings nutrients nearer to the surface to enhance phytoplankton productivity and, in turn, increase coastal fish abundance and growth (Okano et al., 2011).

The complete use of artificial reefs has yet to be fully explored. Interestingly, at least one company (www.eternalreefs.com) makes use of artificial reefs as human burial sites.

Concepts

The general idea behind deploying artificial reefs to enhance fisheries is related to either one or both of the two assumptions long argued by fisheries scientists. These assumptions are that artificial reefs attract fishery resources to a site, or they increase the productivity of fisheries resources. Clearly both concepts are viable and each can have utility in fisheries management. To date, however, both these assumptions are recognized, but little scientific testing has been done to allow proper rejection of either hypothesis in the attraction versus production debate (Bortone, 2006; Bortone, 2011).

Bohnsack (1989) diagrammatically summarized the general perspective of the attraction and production perspectives based on observations and published research. His perspective was that attraction and production were either ends of a continuum with regard to several attributes. In summary, artificial reefs that functioned more for production than attraction had low reef availability and low fishing intensity. They were inhabited by species that were habitat limited (as opposed to recruitment limited), more reef dependent, and, behaviorally, more territorial, demersal, and with high site affinity (philopatry). Polovina (1991) further clarified the attraction/production argument relative to fisheries. He indicated that if artificial reefs merely served to concentrate fish in an area, then the same biomass could be caught with less effort. If the artificial reef attracted fish from other areas, then fishing yield could increase as long as fishing effort increased. Lastly, if artificial reefs increased the carrying capacity of an area, then both total biomass and exploitable biomass should occur.

The presumption by both Bohnsack (1989) and Polovina (1991) is that attraction is the opposite of production. Bortone (2008) proposed that attraction and production were both at play in the response species made to the presence of artificial reef structure. He reasoned that some species could be both attracted to structure and that, for some species, the carry capacity of an area could also allow an increase in biomass as well (e.g., octopus and spiny lobster). Oppositely, some species are neither attracted nor do their populations increase because of the presence of artificial reefs (e.g., species that show no affinity for reefs). Most certainly, a variety of species show varying degrees of attraction to artificial reefs with varying degrees of biomass facilitation because of the reef's presence.

The conundrum might be resolved if resource managers could determine the features of target species that are enhanced by deploying artificial reefs. This means that some species and their associated fisheries may, indeed, benefit from the deployment of an artificial reef, in terms of both attraction and production, while others may not. This implies that, at least if fisheries enhancement is the goal of an artificial reef deployment, each reef should have an objective directed toward a particular species and its life history feature that can be enhanced because of the reef.

Special features of estuarine artificial reefs

While there has been some attention given to the deployment of artificial reefs in estuaries (e.g., Bortone et al., 1994; Chapman and Clynick, 2006), they have received only passing attention to date, although artificial reefs have been used as mitigation in estuaries (Foster et al., 1994). It should be noted that the principles applied to artificial reefs in other aquatic systems are probably no different when applied to estuaries. Generally, diversity issues are less important in estuaries, and changes in salinity, tidal flow, and turbidity add extra dimensions when considering the results of artificial reef deployments. Their more recent usage in estuaries has been via the deployment of oyster reefs (Coen and Luckenbach, 2000). This is a popular estuarine enhancement activity conducted by many resource managers. The extrapolation of deploying oyster reefs as consideration of an estuarine artificial reef should not be overlooked by artificial reef researchers.

Current investigations

Research on artificial reefs continues, but, as Bortone (2011) warned, unless clear objectives are included in these investigative efforts, the resolution of the attraction versus production argument will remain elusive. Particularly disconcerting is the feature that current research results are unable to answer many of the questions resource managers face. While artificial reefs have long been touted as offering a solution to many fisheries management issues, their lack of specific prescription in management will continue to exclude artificial reefs from the proverbial "managers toolbox" until these and many other issues associated with artificial reefs are programmatically resolved.

Gaps in current knowledge

Bohnsack and Sutherland (1985) and Bortone (2006, 2001) presented arguments for new directions in artificial reef research. Each of these reviews indicated the overriding gap in artificial reef research is the lack of application of artificial reefs as a reliable and predictable option for natural resource managers. Resolution of the attraction/production hypotheses plays a prominent role in resolving this issue. More importantly, however, is the need to determine the life history "bottlenecks" that are likely to be relieved by the deployment of a reef. One example of how the future might look with regard to artificial reefs in fisheries can be seen in the example off South Korea (Kim et al., 2011). In this study, artificial reefs were deployed to enhance spawning success in a bottom-dwelling fish species. The reef deployment, coupled with fishing restrictions, is part of a long-term plan to increase a depleted fish stock. Similar investigations may give purposeful direction to future artificial reef designs and deployments.

Summary

Objects of various materials, shapes, and dimensions have been deployed in nearly every aquatic environment in all parts of the world to serve as artificial reefs. Few artificial reefs have been deployed in estuaries. Nevertheless, oyster shell material, deployed to improve oyster settlement can be considered an artificial reef or at least an artificial substrate that behaves similar to other objects deployed as artificial reefs. Artificial reefs have received considerable attention among aquatic scientists and natural resource managers to facilitate fisheries. However, there remain several issues relative to the attraction versus production nature of artificial reefs that need to be resolved before they become part of a regularly prescribed option for estuarine resource management.

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Cross-references

Fringing Reef
Oyster Reef

AUTOTROPHIC

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Definition

Autotrophic organisms use an inorganic carbon compound for their sole carbon source (Okafar, 2011). The corresponding pathways of carbon metabolism are also referred to as autotrophic.

Etymology: from Greek αυτός, self, and Greek τροφή, nourishment.

The carbon sources of autotrophic metabolism comprise carbon dioxide (CO_2), carbon monoxide (CO), and carbon disulfide (CS_2). A majority of autotrophic organisms rely on CO_2 to cover their demand of cellular carbon. Examples include plants, algae, and photosynthetic bacteria, as well as a broad variety of chemotrophic microorganisms. CO-dependent autotrophs are metabolically, ecologically, and phylogenetically diverse (King and Weber, 2007). They convert CO to CO_2 and sometimes methane (CH_4) or acetate (Sokolova et al., 2009; Techtmann et al., 2009). Comparatively few bacteria are known to use CS_2 in their autotrophic metabolism (Cox et al., 2013).

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Cross-references

Heterotrophic
Macroalgae
Microphytobenthos
Phytoplankton

B

BACK DUNE

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Synonyms

Coastal strand; Rear dune; Secondary dune

Definition

Back dune is a generic term for established dunes in a coastal setting that lie detached from the shoreline by other dunes referred to as foredunes (Salm et al., 2000; Hansen et al., 2002; Hansen et al., 2010; West, 2004). The location of the back dunes behind the foredunes generally offers them protection from the direct effects of onshore winds such that the deposition of new sediment or erosion by wind is often minimal (Timmons et al., 2007). As a result, soils may develop on the surface of the back dunes and vegetal communities usually flourish. Blowouts may develop on back dunes if the vegetation on the dunes is disrupted naturally or by human activity.

Origin: The term back dune has no morphogenetic connotations. Hence, any dune shape formed by any process could be described as a back dune. In many cases back dunes are former foredunes and parabolic dunes that become stable. Thus, the term back dune is more of a descriptive term for an environment of occurrence than a genetic term.

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BACKBARRIER

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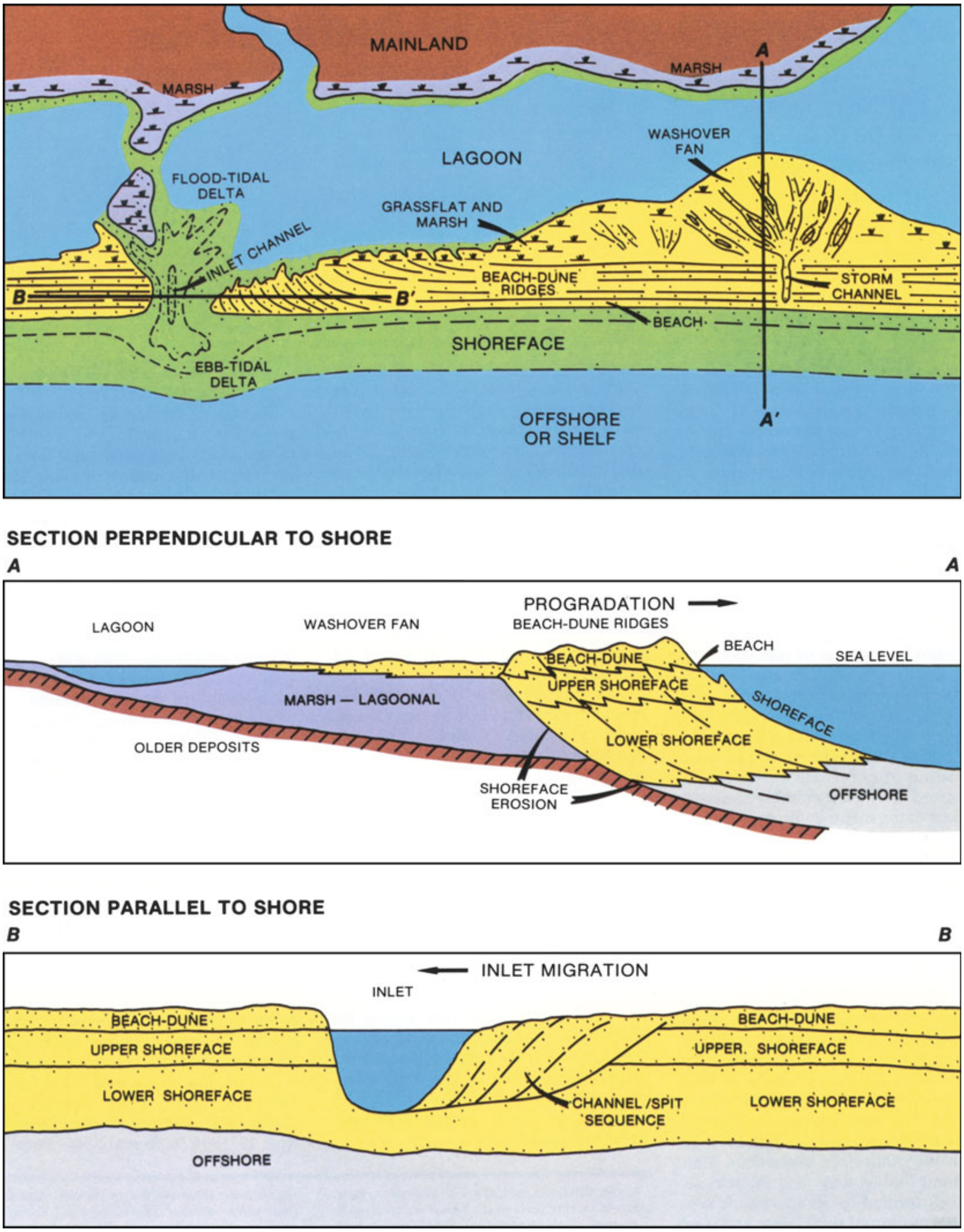
Definition

The backbarrier complex lies between the landward side of a barrier island and the mainland. It encompasses a suite of subaerial, intertidal, and subaqueous depositional environments. The preservation potential for some segments of the backbarrier complex, specifically inlet and flood-tidal delta deposits, is high. Such sediments comprise a large portion of ancient clastic coastal deposits.

Introduction

The backbarrier is a sedimentary environment dominated by fine sand and mud, although significant amounts of coarser sediment may occur locally (Howard and Frey, 1985). Large-scale planar and trough cross-bedding is common, along with graded beds and sand-mud interbedding.

M.J. Kennish (ed.), *Encyclopedia of Estuaries*, DOI 10.1007/978-94-017-8801-4,



Backbarrier, Figure 1 Barrier island complex, showing the subenvironments of the backbarrier system (McCubbin, 1982).

Bioturbation is common, along with burrows. The components of a backbarrier complex (Figure 1) may include *coastal lagoons*, which are shallow basins lying between the mainland coast and the landward side of a barrier island; *flood-tidal deltas*, which are formed through deposition on the lagoon floor by flood-tidal currents flowing through a coastal inlet; *washover fans*, which are represented by subaerial sheets of coarse sediment which have been carried landward through storm-created gaps in the barrier front; *intertidal flats*, which are the segments of the backbarrier complex lying within mean tide range; and *intertidal marshes*, consisting of the portion of the intertidal region on which salt marsh vegetation is able to grow (McCubbin, 1982; Howard and Frey, 1985; Friedman et al., 1992; Davis and Fitzgerald, 2004).

Coastal lagoons

Lagoons comprise the open-water areas between the barrier proper and the mainland beach. The occurrence of overwash builds out the barrier platform, reducing the tidal prism and altering circulation within the lagoon (Cooper, 1994). Howard and Frey (1985) characterized lagoons as salt marsh estuaries, driven by tidal circulation, as opposed to riverine estuaries, which have a freshwater river source at their head. They noted that the sedimentary characteristics of the two environments were similar and therefore difficult to distinguish in the rock record. Coastal lagoon sediments are composed of silt and clay and are extensively bioturbated (Oertel (1985). Lagoons can be characterized as open-water lagoons or expandable lagoons. Open-water lagoons have a relatively constant water surface area. The surface area of expandable lagoons may vary by as much as 50 % between spring low and high tides. The latter can evolve into the former, provided that the rate of submergence due to sea-level rise exceeds the rate of sediment accretion (Oertel, 1985).

Flood-tidal deltas

Flood-tidal deltas are formed by tidal sediments deposited landward of an inlet mouth. As inlet channels fill and inlets migrate, flood-tidal deltas become inactive and eventually become part of the barrier (Carrasco et al., 2008). This process is one of the principal means by which the backbarrier environment builds outward (Godfrey and Godfrey, 1974). One of the most common backbarrier sedimentary sequences fines upward from coarser inlet deposits to fine-grained flood-tidal delta sands to salt marsh. These sequences comprise a major part of the barrier facies and account for up to half of the Holocene barrier sediment (Moslow and Tye, 1985).

Washover fans

Washover fans are the accumulated product of short-term depositional events during storms that breach the barrier front. Overwash, which affects both the width and height of the barrier platform, is a major control on backbarrier development. When a storm event causes marine water to reach the lagoon, lenticular washover fans are deposited on the backbarrier margin (Carter, 1988). The washover sediments are the result of erosion of barrier dune and beach environments and overlie former salt marsh (Schwartz, 1981). The importance of overwash as part of the barrier lithesome depends on the bathymetry of the foreshore (Ritchie and Penland, 1988), wave conditions (Fisher et al., 1974), and elevation of backbarrier beaches (Morton and Sallenger, 2003). Overwash can have either a positive or negative effect on backbarrier evolution, depending on the frequency and intensity of overwash events (Godfrey and Godfrey, 1974).

Sedgwick and Davis (2003) described the characteristics of washover facies. Washover beds are typically landward-dipping plane beds of well-sorted sand. Shell beds and heavy mineral laminae are often interbedded with sand layers. Bioturbation and reworking by later events can overprint the record. Washover deposits are often difficult to distinguish from flood-tidal delta sediments. Washover deposits in the stratigraphic record are characterized by (1) landward thinning, (2) occurrence of clean sand deposits within the find-grained backbarrier sediments, and (3) presence of shoreface and backbarrier mollusk shells (Sedgwick and Davis, 2003).

Intertidal flats

Intertidal flats lie at elevations between mean high and mean low tide. They may be thought of as salt marshes lacking in vegetation and provide the substrate upon which salt marshes build. The sediments of intertidal flats consist of interbedded mud and sand, representing cyclic changes in tidal current velocities (Howard and Frey, 1985). Bedding varies from flaser to wavy to lenticular, depending on the relative proportion of sand and mud (Reineck and Wunderlich, 1968).

Intertidal marshes

The backbarrier marsh environment includes grass beds and tidal channels lying within the range of mean tides. Backbarrier marshes generally evolve on tidal flats situated between the tidal channels of an abandoned inlet system (Kraft et al., 1979). Tidal current velocities flowing over tidal marsh surfaces are typically an order of magnitude lower than those observed in tidal channels (Howard and Frey, 1985). Bartholdy et al. (2010) reported that the backbarrier marsh is highly sensitive to the rate of sea-level rise. Continued deposition in the marsh environment requires a positive and constant rate of sea-level change. Sea-level stasis, or an increase in the long-term rate of rise, leads to loss of the marsh. Godfrey and Godfrey (1974) noted that excessive overwash can overcome the ability of the marshes to recover and lead to destruction of the marsh environment. The higher elevations in the salt marsh, however, are dependent on overwash events to supply sediment for accretion (French and Spencer, 1993).

Carrasco et al. (2008) developed an evolutionary model for the backbarrier environment, based on the linear extent of salt marsh development along the backbarrier shoreline versus the length of non-vegetated backbarrier beach. The ratio of salt marsh to beach was found to be related to changes in local hydrodynamic conditions. A decrease in hydrodynamic intensity results in a higher ratio of marsh length to beach length. An increase in hydrodynamic intensity, such as the creation of new overwash pathways, results in a lower ratio. The model can be employed to project future changes in the backbarrier environment.

Summary

Backbarrier sediments are a complex of various interfingering subenvironments. Facies models of the several subenvironments can be useful in identifying barrier facies in the rock record. Delineation of individual backbarrier facies is often difficult due to bioturbation, reactivation, and reworking.

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Cross-references

Back Dune
Barrier Island
Barrier Spits
Beach Processes
Coastal Barriers
Coastal Lagoons
Coastal Landforms
Estuarine Beaches
Estuarine Geomorphology
Intertidal Zonation
Overwash
Saltmarshes
Washover Fans
Washovers

BAR

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Synonyms

Sand bank; Sand bar

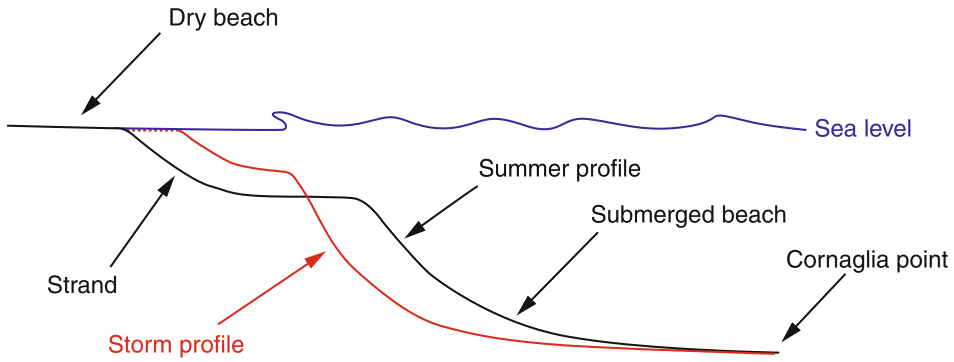
Definition

The term *bar* refers to a step or projection in the cross profile of a beach. While a bar may have slightly different meanings when used by different authors (King, 1972; Finkl, 2004), in all cases the term can be linked to the transformative action of waves when they approach the coastline over a sea bottom that consists of non-cohesive granular sediment.

The concept of a bar is relevant for interpreting data and gaining knowledge of almost all sedimentary coastal formations such as cordons, barrier (sedimentary) islands, hooks, spits, cuspate forelands, and tombolos. For detailed analyses of such formations (Williams, 1982), the Genetic Classification of Simple Coastal Forms (Bores, 1978) is a valuable resource.

Genesis

Water depth gradually decreases as a wave approaches the coast, and mass transport is accentuated because of the asymmetry of the open wave orbital motion. Thus, sea bottom sediments are dragged up toward the breaker line, which generates an increasingly stepped slope toward the shore and carves out a concave profile. Wave motion stops at the breaker line in a tide-free sea, and the cross profile exhibits a geometric discontinuity in that location. This discontinuity is the bar.



Bar, Figure 1 Variability of the beach cross profile and its bar.

As each wave breaks, the orbital energy is converted into kinetic energy over the strand (see Figure 1), where it generates maximum turbulence and stirs up sediments. Then, this energy turns into potential energy on the berm. Finally, the energy is transformed into kinetic energy once again by the falling water, which drags sediments in the offshore direction. Sediments are moved by gravity in the offshore direction from the berm at an increasing speed.

The profile resulting from a monochromatic wave would be a double concave curve with a slope that increases coastward and with the point of discontinuity (bar) at the breaker line. However, wind-generated waves over the sea are irregular, and the associated bars that form are spread out over a wide area. If a sea state lasts long enough, it can carve out a convex-shaped bar in the breaker area that separates the other two concave curves of the cross profile.

Changes over time constantly wash away and carve out new bars. The bars change in size and location depending on the energy and duration of wave action. Hence, bars only consolidate following a certain amount of climate stability and constant wave action. This typically occurs (1) after periods of calm or gentle swells in the summer, where the bar and wave profile prevails, and the berm is more advanced toward the water, and (2) after winter storms, where the bar and storm profile prevails and the berm is more withdrawn landward. The latter bar is then wider, deeper, and farther from the shoreline. The profile of a beach in equilibrium generally displays these two extreme conditions, which vary from 1 year to the next. Additionally, there are coasts where the climate or continuous increases in sediment (hyperstability) lead to profiles with more than one bar (e.g., off the Dutch North Sea islands).

Each sea state requires a volume of sediment over and above a threshold level for bars to form. The absence of a bar on a coast with sediments along the entirety of its cross profile is therefore indicative of instability or erosion.

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Cross-references

Barrier Island
Beach Management
Littoral Cordon
Spit

BARRIER ISLAND

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Synonyms

Barrier

Definition

A barrier island is a coastal landform consisting of an elongated narrow strip of unconsolidated sediment (primarily sand) lying parallel to the mainland coast and being separated from the mainland by a lagoon, salt marsh, or bay.

Introduction

Barrier islands are found worldwide. However, there exist different opinions on the definition of a barrier island. Descriptive definitions of barrier islands, such as the one adopted in this entry, are generally more widely accepted than genetic definitions which consist of quantitative limitations (e.g., Berryhill et al., 1969; Cromwell, 1971). Some researchers define barrier islands as features composed of several major depositional units due to the strong links among these units that are required for their existence.

Barrier Island, Figure 1 Examples of barrier islands in open-ocean (*left*) and fetch-limited (*right*) environments. *Left*, the Dutch Wadden Sea coast; *Right*, the Smith Island in the Chesapeake Bay, USA (Image source: Google Earth 2013).

One example of a systemic definition is given by Oertel (1985) who suggested that a barrier island should be considered as the focal element of a much larger barrier island system, consisting of six major elements: (1) mainland, (2) back-barrier lagoon, (3) inlet and inlet deltas, (4) barrier island, (5) barrier platform, and (6) shoreface. A lack of any one of these elements would result in misuse of the term barrier island.

According to the criteria from Oertel (1985), a recent survey by Stutz and Pilkey (2011) based on global satellite data combined with topographic and navigational charts identified 2,149 individual barrier islands totaling 20,783 km in length, taking up about 10 % of all continental shorelines. In this case only open-ocean barrier islands are taken into account in the survey due to the limitations of the criteria. However, if the criteria are not restricted to the systemic definition but include other islands that meet the descriptive definition as given in this article, another category of barrier islands characterized by sheltered, low-wave energy coastal environments (so-called fetch-limited) would be applicable. Pilkey et al. (2009) reported the existence of more than 15,000 fetch-limited barrier islands developed in the sheltered waters of fjords, bays, lagoons, and behind coral reefs. Due to the absence of driving forces under fair-weather conditions and low sediment availability, development of fetch-limited barrier islands is strongly dependent upon stochastic extreme events (e.g., storms, floods). Fetch-limited barrier islands have a much smaller size than open-ocean barrier islands. They are typically short (~1 km), narrow (some tens of meters), and low lying (mostly less than 3 m above the mean sea level), while open-ocean barrier islands have an average length of 8.8 km and a width of 0.7 km, according to Stutz and Pilkey (2011). Dune ridges higher than 10 m can also develop on some open-ocean barrier islands if aeolian onshore transport is strong and sediment source is abundant (e.g., the Algarve barrier island chains along the south coast of Portugal). Figure 1 shows examples of open-ocean and fetch-limited barrier islands.

Origin of barrier islands

As shown in Figure 1, barrier islands normally occur in chains, which can be found in quite different climatic environments (e.g., from Arctic to tropical zones), suggesting that they are relatively flexible and can form and sustain in a variety of environmental settings. For more than 150 years, coastal researchers have investigated the origin of barrier islands. Numerous theories have been developed to explain their formation and development. By the end of the nineteenth century, three original hypotheses were available. De Beaumont (1845) suggested that barrier islands, such as those found in the North Sea and the Gulf of Mexico, were formed by the emergence of submarine bars. On a low-gradient coast, waves tend to break away from the shoreline enabling the buildup of submerged bars away from the coast, which then gradually grow in size and emerge due to the impacts of waves and aeolian transport. Gilbert (1885) suggested that barrier islands can form from a spit generated by longshore drift. During storms, the spit is breached, creating inlets that divide the spit into a series of islands. McGee (1890) proposed that barrier islands are produced by drowning of coastal ridges during sea-level rise or tectonic subsidence. Since then, there has been considerable debate (e.g., Hoyt, 1967; Fisher, 1968; Otvos, 1981) over these three hypotheses. Until recent decades (e.g., Schwartz, 1973), it has been determined that these three hypotheses can explain the formation of different types of barrier islands, but no single one can fully explain the development of all barriers distributed worldwide. More and more studies (e.g., Schwartz, 1971; Hayes, 1979; Leatherman, 1979; Leatherman, 1985) have shown that the formation and development of barrier islands are a result of multiple processes.

Boundary constraints

Although the environmental conditions required for formation of barrier islands are relatively flexible, there still exist some boundary constraints. According to the statistics from Stutz and Pilkey (2011), barrier islands are most abundant (~63 % of the total) on tectonically stable, trailing edge continental margins as such environments provide favorable boundary conditions (e.g., abundant sediment supply, small ratio of tidal range to mean wave height) for the formation of barrier islands. Of the remaining barrier islands, ~21 % are located on marginal seas, and only ~16 % are found on collision margins. Most (~58 %) of the barrier islands existing on collision margins are developed on delta lobes favored by a low-gradient shoreface produced by abundant riverine sediment input; the rest are located on wide coastal plains. Barrier islands rarely form on narrow continental shelves with an upper shoreface slope larger than 0.8°, in which sediment tends to move offshore rather than accumulating onshore.

Another significant boundary factor influencing the formation and development of barrier islands is sea-level change. A stable sea level is a prerequisite for the formation of barrier islands. Most barrier islands are quite young, being formed during the last ~6,000 years when the global sea level became relatively stable with only minor fluctuations. A stable sea level with small rates of change (within millimeters per year) in the mid- to late Holocene restricts tides and wave actions to a small-range coastal area (i.e., hydrodynamically active zone). Sediment transport within this area became increasingly important to shape the modern coastline. Driven by wave and aeolian processes, an excess of sediment supply to a local accommodation zone would eventually build up new land above the water surface. Holocene barrier spits and islands present such examples. Holocene barrier islands are low-lying structures made of unconsolidated sediment, with the highest part at the dune crests, which is normally only meters above the water level. Thus they are quite vulnerable to high water-stand impacts induced by storms or floods. Without sufficient sediment supply to compensate the increased accommodation zone, continuous sea-level rise would cause a barrier island to shrink and migrate landward.

Besides a sufficient sediment supply to feed the formation of a barrier island, the “quality” of the sediment supply is also critical for the fate of the island. Sediment supply with a larger proportion of sand and coarse material is able to sustain stronger hydrodynamic impacts than fine sediment such as mud and clay. Thus, sandy substrate and shoreface are more durable than a muddy one to maintain a barrier island. The wave-tide regime is also an important factor influencing the morphogenesis of barrier islands. Beaches and barrier islands are products of wave action. They develop most easily on wave-dominated coasts with small to moderate tidal range. Only ~12 % of barrier islands develop in tide-dominated regions (with the ratio of mean tidal amplitude to mean annual wave height generally larger than three according to Davis and Hayes, 1984), and they are rarely found in areas with a tidal range larger than 4 m.

Barrier island morphodynamics

Among different types of coastal landforms, barrier islands have the most variable morphology. They are constantly shaped by winds, tides, and waves and, on a longer time scale, can shift landward or seaward due to oscillations of sea level and variations in the sediment supply (Masetti et al., 2008).

Depending on the relative importance of waves to tides in determining the coastal morphology, three types of coastal environments can be classified: wave dominated, tide dominated, and mixed energy. In wave-dominated coasts, barrier islands are elongate and narrow due to the impact of longshore drift. Inlets produced by tides or storm breaching migrate fast for the same reason. Washover features are prominent, and flood deltas are well developed but ebb deltas are small or nonexistent (Hayes, 1979). Along with an increase of tidal effect, inlets play a more significant role in shaping the barrier island morphology. Substantial ebb deltas can develop, and barrier islands become shorter and wider as a result. As tidal range increases, these features become more prominent. When the tidal range is high enough and overwhelms the wave effects, barrier islands cannot develop and inlet deltas are confined to elongated stringers oriented with the dominant tidal currents.

In addition to tide and wave actions, development of barrier islands is also affected by other processes (e.g., stochastic extreme events, sea-level change, tectonic movements, and fluvial input). Barrier islands evolve and migrate parallel or normal to the mainland in response to these processes. The shore-normal evolution of barrier islands corresponds to two types of behavior: namely, regression and transgression, respectively. Barrier transgression refers to an onshore migration of the landform and an overlapping of deeper water sediment over shallower lagoon deposits. Leatherman (1979) summarized three main processes controlling barrier island transgression, which, in the order of importance, are inlet dynamics, overwash, and aeolian transport. In some areas with a thick and compressible substrate (e.g., the Virginia barrier coast), auto-compaction also contributes to the barrier island transgression (Leatherman, 1985). In response to the increased impacts of these processes induced by an eustatic sea-level rise, three modes have been proposed to describe a subsequent evolution of a barrier island: (1) a continuous landward migration across the underlying substrate to higher elevations; (2) a disintegration of the island due to insufficient sediment supply and backshore relief to sustain inundation during stochastic extreme events; and (3) an in-place drowning which turns the island into a submarine deposit body. Although there exist some cases to support the latter two modes, the most common mode of barrier island transgression is the continuous landward migration through the combined effects of shoreface erosion, overwash, and inlet floods (including storm breaching). Through a continuous "rolling over" itself, the barrier island eventually merges with the mainland, with its upper layers of sediment eroded and recycled.

Barrier regression refers to an offshore expansion of the landform and shoaling growth. It is a result of an excess of sediment supply to the island. Sediment supply mainly comes from three sources: river input, longshore drift, and onshore migration of submarine sandbars. In the process of barrier island regression, the outer (ocean-ward) shoreline progrades seaward, while the inner (lagoon-ward) shoreline remains relatively stable, forming a wide low-lying plain characterized by multiple dune ridges, normally with the most seaward foredune ridges possessing the highest elevation. Such high foredune ridges may prevent overwash and thus help to protect the island from storm erosion; however, meanwhile they also block the transport of sediment to the backshore and may accelerate the erosion on the inner shoreline during eustatic sea-level rise over the long term. Accompanied by a decreased sediment supply, this may lead to a switch of the barrier evolution to a transgression phase.

Numerical modeling

Due to high sensitivity to boundary conditions, natural barrier islands serve as an ideal laboratory for numerical studies of multi-scale physical processes on the coastal morphological evolution. They can also be studied as proxies of long-term climate change (Zhang et al., 2014).

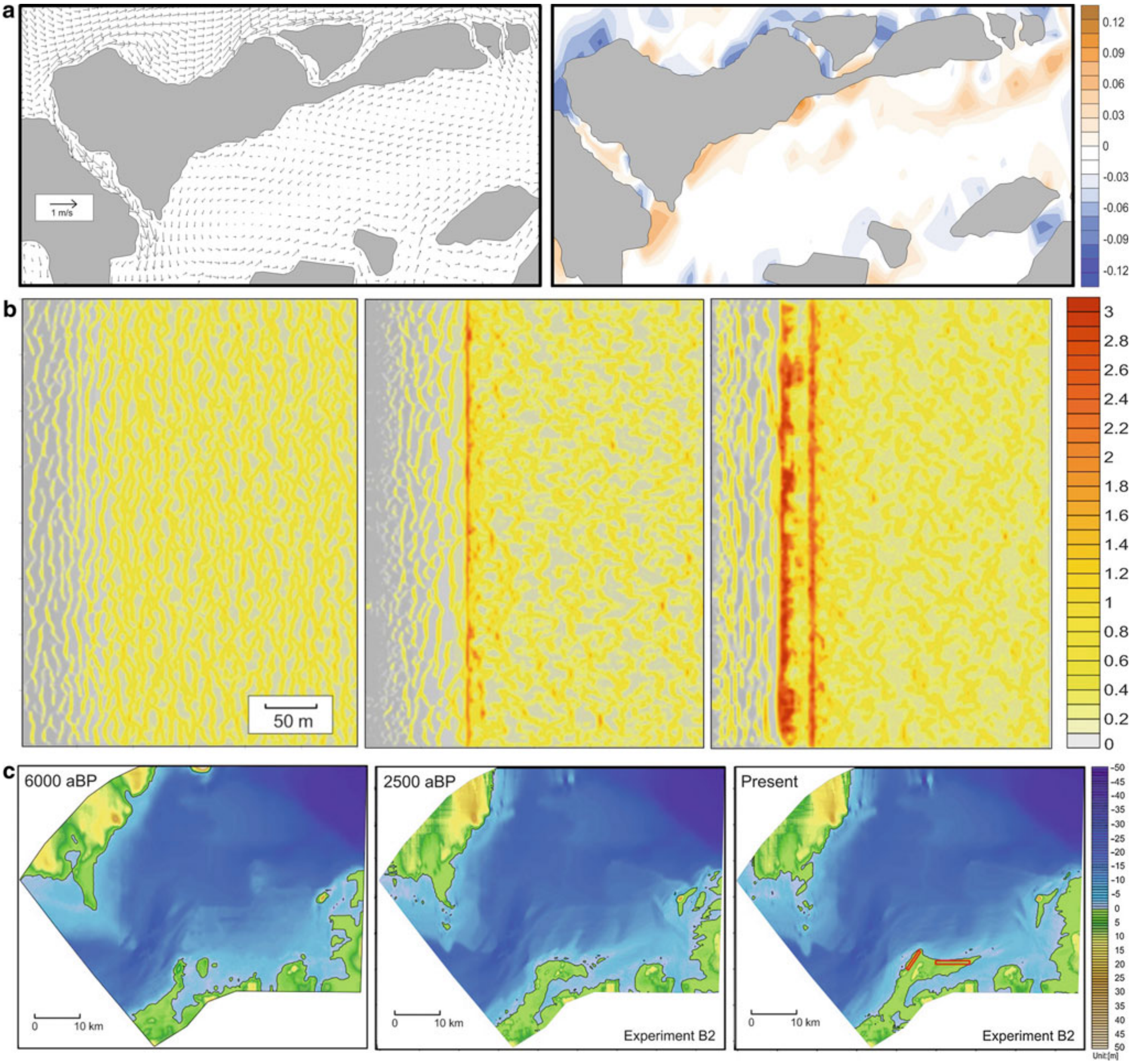
Morphogenesis and evolution of barrier islands are complicated due to the influence of many processes occurring at different temporal and spatial scales as discussed previously. Due to difficulties resolving all relevant processes and their interplay in an integrated numerical model, simplifications are usually used in mathematical descriptions of these processes and their corresponding scales.

The most common numerical models available for study of barrier island evolution are 2-dimensional vertical (2DV) cross-shore profile models. In these models, morphological response of a cross-shore coastal profile to actions of several key processes is used to represent the evolution of the whole barrier island. The coastal profile is selected in such a way that it should be able to represent typical characteristics of the barrier island and its adjacent environments. The profile starts from a high terrestrial point at the mainland and extends seaward to an offshore closure point. Outer areas beyond these two points are presumed unchanged and do not impose any effect on the barrier island system during the time span of interest. After a setup of the initial profile shape and other parameters (e.g., sediment composition, grain size, substrate lithology), response of the profile shape and underlying stratigraphy to influences of different processes is calculated through a set of equations. Depending on the equations adopted, 2DV cross-shore profile models can be further classified into two different types: process based and behavior oriented. Process-based models apply a set of differential equations to describe the wave transformation, sediment transport, and subsequent bed elevation change on the profile. Impacts of storm surge, eustatic sea-level change, and tectonic movement are implemented in the equations through a parameterization of boundary conditions (i.e., incoming wave properties and water level). Examples of process-based 2DV models can be found in Masetti et al. (2008), Rosati et al. (2010), and Zhang et al. (2013). On the contrary, behavior-oriented models (e.g., Roy et al., 1994; Cowell et al., 1995; Storms et al., 2002; Stolper et al., 2005; Moore et al., 2010) describe the profile change by a set of empirical functions of changes of sediment supply, sea level, and shoreface geometry, without simulating the detailed processes involved in sediment transport.

The validity of 2DV cross-shore profile models is based on three pre-assumptions: (1) a zero net sediment exchange at the boundary (thus sediment is conserved in the profile); (2) evolution of the shoreface part of the profile evolves toward a predefined shape (the so-called equilibrium), which is determined by the grain size of the shoreface sediment and the mean wave climate (e.g., Bruun, 1962; Dean, 1991; Dean, 1997); and (3) alongshore uniformity of offshore wave parameters and nearshore isobaths along the coastline (thus the gradient of longshore sediment transport rate is zero and does not affect the profile change).

2DV cross-shore profile models have proven to be more useful in providing detailed insights into the fundamental driving mechanisms of barrier island development than conceptual models. However, one should always keep in mind the limitations of validity which may hinder application of a 2DV model to a real case. Another factor affecting the reliability of a 2DV model is an exclusion of inlet effects, which are most critical in controlling barrier island morphodynamics according to Leatherman (1979). An extension of an individual profile to an area might overcome these limitations; however, this requires much greater effort in bridging the different scales that are involved in barrier island morphodynamics. Development of such models is still at an early stage. An example of such models is presented by Zhang et al. (2012, 2014).

A hybrid and parallel coupling of process-based and behavior-oriented modules provides a way to resolve the relevant processes at their corresponding scales with an affordable computational expense. In the model, wave processes (propagation, transformation, refraction, and breaking), currents, and subaqueous suspended sediment transport are solved in process-based modules, while subaerial aeolian transport, bed-load transport, and land-sea transition processes (e.g., cliff erosion) are simulated either in behavior-oriented manners or by cellular automata approach. The model was applied to investigate the morphogenesis and evolution of a Holocene barrier island



Barrier Island, Figure 2 Processes and morphological change at different scales during the development of a barrier island (Darss-Zingst) at the southern Baltic Sea. (**a**) Storm breaching (*Left*) at the barrier spit and subsequent bed elevation change (*Right*); (**b**) development of foredunes at the back-beach area; (**c**) simulated morphological evolution of the barrier island at different stages. The foredune planes developed on the island are indicated by frames (Images are modified from Zhang et al. (2014). Unit is meter in all images).

(Darss-Zingst) at the southern Baltic Sea. Although the gap between the simulated morphology and the real situation as seen today is remarkable, the model proved to be able to reproduce the main morphological features of the barrier island system, e.g., the development of two barrier spits, foredune plains and the inner lagoon, and major driving mechanisms (storms, inlet erosion and deposition, aeolian transport, littoral drifts, and cliff erosion) for the island formation. Figure 2 shows some simulation results.

Summary

Morphogenesis and evolution of barrier islands are complicated processes. It is clear that the formation of a barrier island is the result of multiple processes and no simple hierarchical relationship can be deduced among the processes in influencing barrier island development. Among different processes that may affect the morphological development of a barrier island, some are found to be of critical importance and act as universal boundary constraints. The history of tectonics and eustatic sea-level change seems largely to determine whether a barrier island can be formed. With favorable tectonics and sea-level change for barrier island formation, a combination of shoreface slope, wave-tide regime, and sediment source determines how a barrier island is formed. Wind-wave climate, as well as the rate and composition of sediment supply, subsequently affects the migration and development of the island. In most areas, vegetation properties (e.g., species, coverage) are important in shaping the island morphology. In areas with a thick and compressible substrate, local consolidation through self-loading of underlying substrate also plays a role in barrier island evolution.

Numerical modeling provides a way to quantify the effects of different processes on barrier island morphogenesis and evolution. However, one should always be aware of the limitations when constructing a model for a specific research object, and simulation results should be carefully interpreted. Much progress is still needed to develop robust models for better understanding of the origin and development of barrier islands.

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Cross-references

Barrier Spits
Spit

BARRIER SPITS

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Synonyms

Barrier; Spits

Definition

Barrier spits are long narrow strips of depositional bodies emerging from water (Evans, 1942), with one end attached to a coast that serves as the source of sediment supply (proximal end) and the other end jutting into open water (distal end), forming a shelter for its inner water.

Introduction

Offshore waves normally approach the surf zone of a coast in an oblique angle. A combination of shore-oblique swash caused by the incoming waves and shore-normal backwash caused by gravity creates a longshore drift, which is further strengthened by longshore currents generated by wave breaking. Sediment is entrained by strong turbulence induced by wave breaking and transported down-drift along the coastline by longshore currents. Longshore sediment transport rate remains constant if there exists a uniformity of waves and nearshore isobaths along the coastline (USACE, 1984). Net deposition occurs where the longshore uniformity is broken by a decrease of the wave energy. This is normally caused by a deepening of the bathymetry or a change of the coastline orientation. In the latter case, the boundary constraint of the longshore currents by the coastline no longer exists, and the currents are veered by a barotropic pressure induced by the wave radiation stress. On the side to which the currents are veered, turbulence is dissipated by free calm water and no longer able to entrain the full load. Much of the sediment is deposited as a result, forming a submerged bar. This submerged bar subsequently acts to maintain the original direction of the longshore currents and on the other hand serves as a reservoir for sediment deposit. Deposition on the submerged bar will not stop until a uniformity of waves and nearshore isobaths is again built up. Eventually an above-water spit is developed and elongated by this process. Terrestrial onshore aeolian transport, which builds up foredunes on the berm of the newly formed beach, plays a key role in stabilizing the spit and allows a further development of the spit. The spit becomes a barrier spit when it is long enough to provide a shelter for its inner water. As foredunes are a common feature on a barrier spit, they serve as useful records for historical environmental change (Tamura, 2012).

Three basic preconditions have to be fulfilled for the formation of a barrier spit:

1. A littoral drift to provide continuous sediment supply
2. A change in the coastline orientation (i.e., a turning point) that is significant enough to remove the boundary constraint of the longshore currents
3. A weak offshore transport at the turning point to enable a major part of the deposited sediment remaining on-site.

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BEACH MANAGEMENT

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Definition

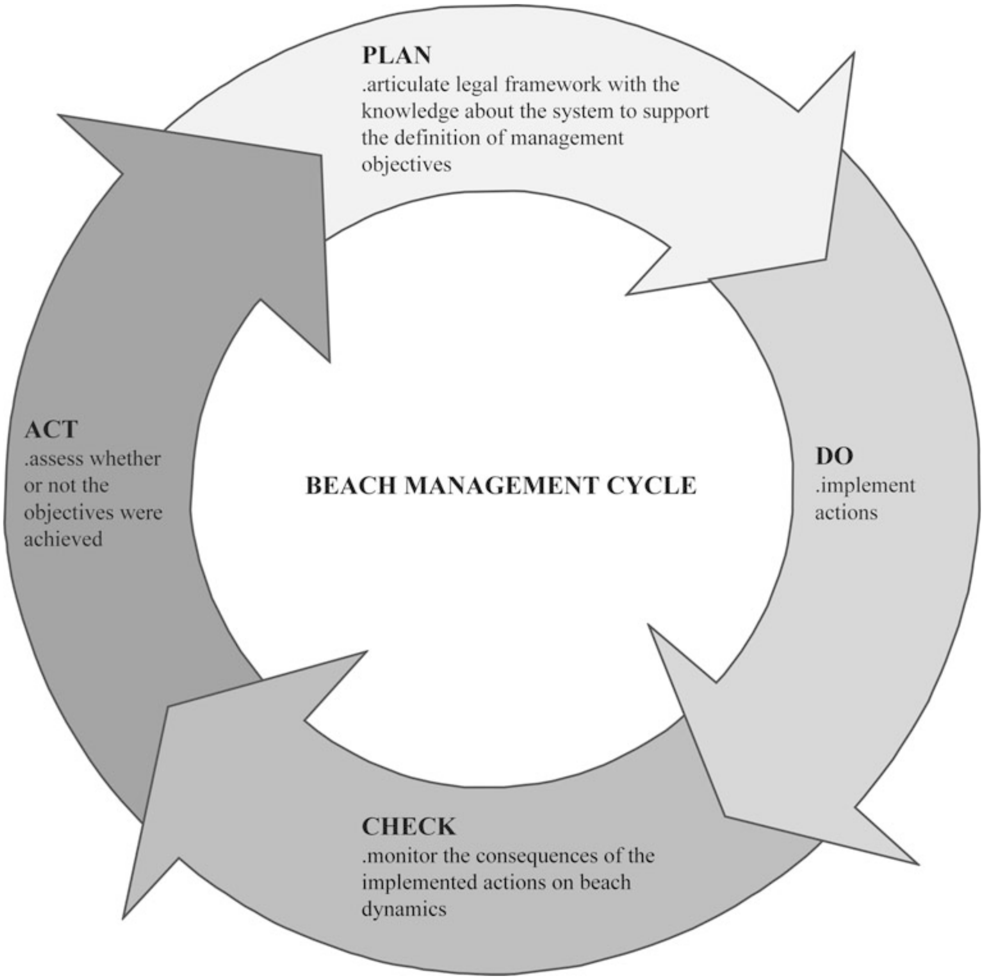
Beach management is the act of organizing and coordinating efforts to identify and implement the optimal use of means to accomplish an end for a specific beach.

Introduction

The process of managing implies the existence of objectives and is used as a technique for achieving an end. Managing also implies knowing the system being investigated, and this constitutes a tremendous challenge in highly dynamic systems, such as a beach, one of the most mutable environments in the world, where morphologic variations can occur on time scales from a few seconds to thousands of years and more and on space scales from meters – or even less – to thousands of kilometers.

In any management activity, four steps are mandatory: plan, do, check, and act (approach known as the PDCA or Deming cycle). *Plan* implies the establishment of management objectives, *do* corresponds to the implementation of the objectives, *check* implies to the objectives implemented and compare them against the expected results (targets or goals from the *plan*) to ascertain the differences, and *act* may imply corrective actions on differences between actual and planned results.

In beach management, the same approach should be considered with the necessary adaptations due to the particularities of the beach environment. In fact, the beach management approaches described in the literature are, in general, implicit deviations of the PDCA cycle (e.g., Micallef and Williams, 2002; Drake, 2010).



Beach Management, Figure 1 Beach management cycle.

Beach management cycle

The beach management cycle is a graphical way, based on the PDCA approach, of synthesizing the key elements in beach management (Figure 1).

First, it is necessary to plan and to establish a management strategy and actions. This step implies to articulate the legal framework, the knowledge of beach managers, and the scientific community concerning beach processes and response, with the strategic and operational management objectives for a specific beach. While strategic objectives provide the long-term context for management and are based on a vision of the natural and the socioeconomic systems (e.g., sustainable development of the beach area), the operational objectives implement the strategic objective (e.g., determine the ecological beach carrying capacity) (van Koningsveld, 2003).

After the actions are established, they should be implemented according to previously established guidelines elaborated in the planning phase.

Monitoring the beach dynamics and response to the implemented actions is the next step. Did things happen according to the plan? Did the system respond as expected? Indicators (quantitative/qualitative statements or measured/observed parameters) should be used in order to support beach monitoring since their main functions are to simplify the information, quantify the target system, and facilitate the communication process between different beach stakeholders (e.g., beach managers and scientific community) (UNESCO, 2006).

The beach management cycle is then completed by evaluating whether the initial objectives were achieved and eventually acknowledging the need to review the original objectives in the context of changing pressures (e.g., climate change) to reflect changes in legislation or good practice (Drake, 2010).

The beach management cycle should be rooted in the integrated coastal zone management philosophy (Cicin-Sain and Knecht, 1998), thus being a dynamic, multidisciplinary, and iterative process aiming to promote sustainable management of the beach area. It seeks over the long term to balance environmental, economic, social, and cultural objectives while acknowledging the specificities of this environment.

Estuarine beaches

Although the beach management cycle is a conceptual approach and therefore applies to all types of beaches, estuarine beaches constitute a singular feature that should be highlighted. Usually, these beaches are subsumed under existing policies to manage ocean beaches, and their specific physical and biological processes and intrinsic values are not always attended in management (Nordstrom, 1992). Estuarine beaches differ from their ocean counterparts in terms of physical structure, social perception, intrinsic values, human use levels, and types of development pressure (Nordstrom, 1992). For this reason, estuarine beach management should be conducted with a thorough understanding of their peculiar dynamics.

Conclusions

Beach management can be described as the act of organizing and coordinating efforts to archive a desired goal for a specific beach. In beach management, four key elements are identified based on the PDCA cycle approach: plan (establish a management strategy and actions), do (implement management actions), check (monitor), and act (assess the degree of achievement of the previous objectives and eventually rethink the initial strategy). For proper beach management, all of these steps should be integrated with scientific knowledge of beach dynamics.

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Cross-references

Beach Processes
Coastal Indicators
Estuarine Beaches

BEACH PROCESSES

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Definition

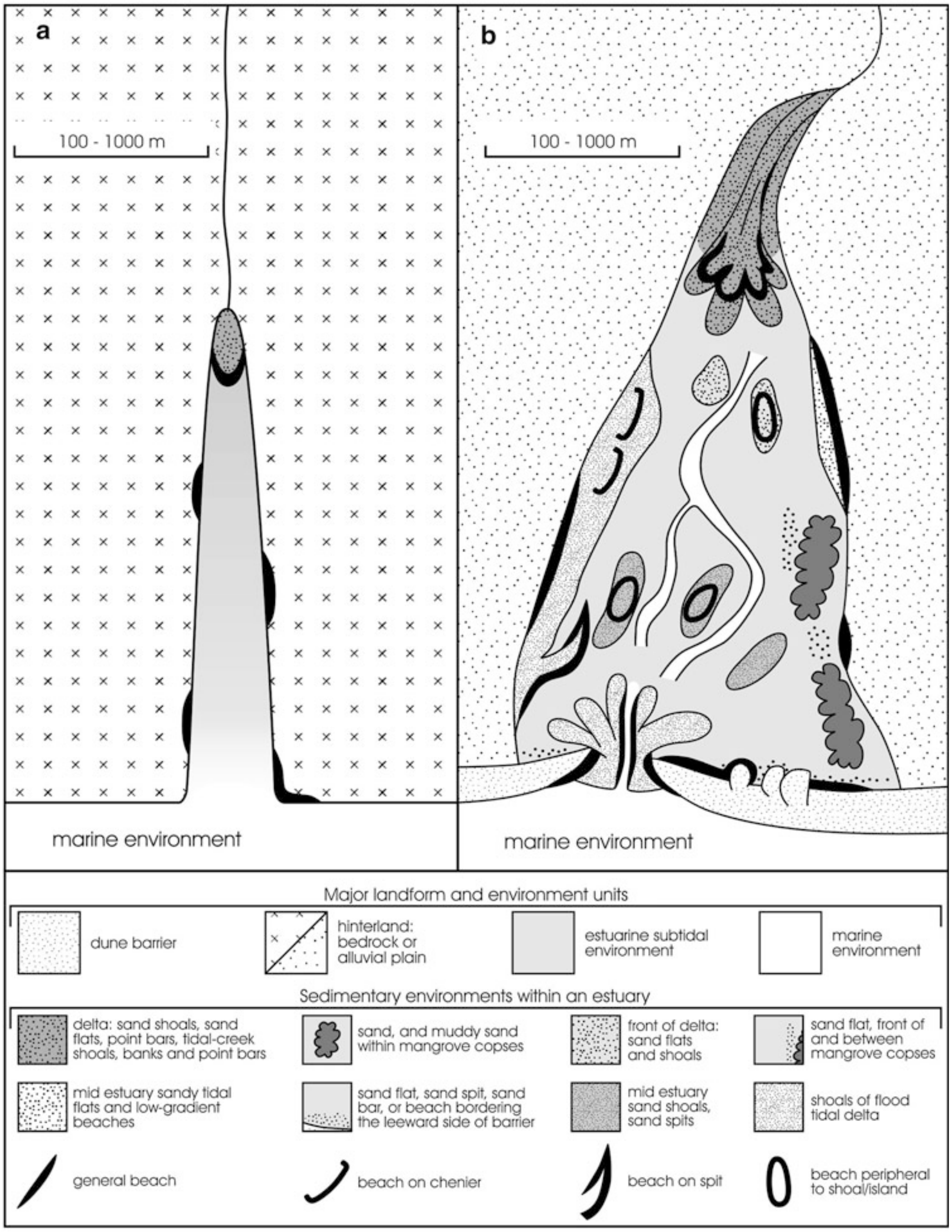
The physical, biological, and chemical processes operating on the surface and shallow subsurface of a beach, resulting in stratigraphic, granulometric, and sediment compositional variations, construction of physical sedimentary structures and biogenic structures, authigenic/diagenetic mineral responses, and biogenic mineral products.

Introduction

A beach can be described as a shoreline that has formed and has been reworked by waves or tides, and that is usually underlain by sand or gravel, and lacking a bare rocky surface (modified from Bates and Jackson, 1987). Beaches largely encompass the tidal interval, but can extend to a limited distance inland, either to a definite change in material or physiographic form (such as a cliff) or to the line of permanent vegetation (usually the effective limit of the highest storm waves). Beaches form in many shore environments, e.g., along mainland coasts fronting an open ocean, small seas, embayments, bays, estuaries peripheral to coral reef islands and volcanic islands, and lakes. This contribution focuses only on the beaches occurring within estuaries and extends, to a limited extent, to the open marine coast (Figure 1).

In tropical regions, some sandy beaches may be inhabited by mangroves, but not to the extent that the beach is fully covered by mangroves. Mangroves, if they inhabit sandy beaches, are generally in the mid-tidal areas, and the high-tidal part of the beach is vegetation-free.

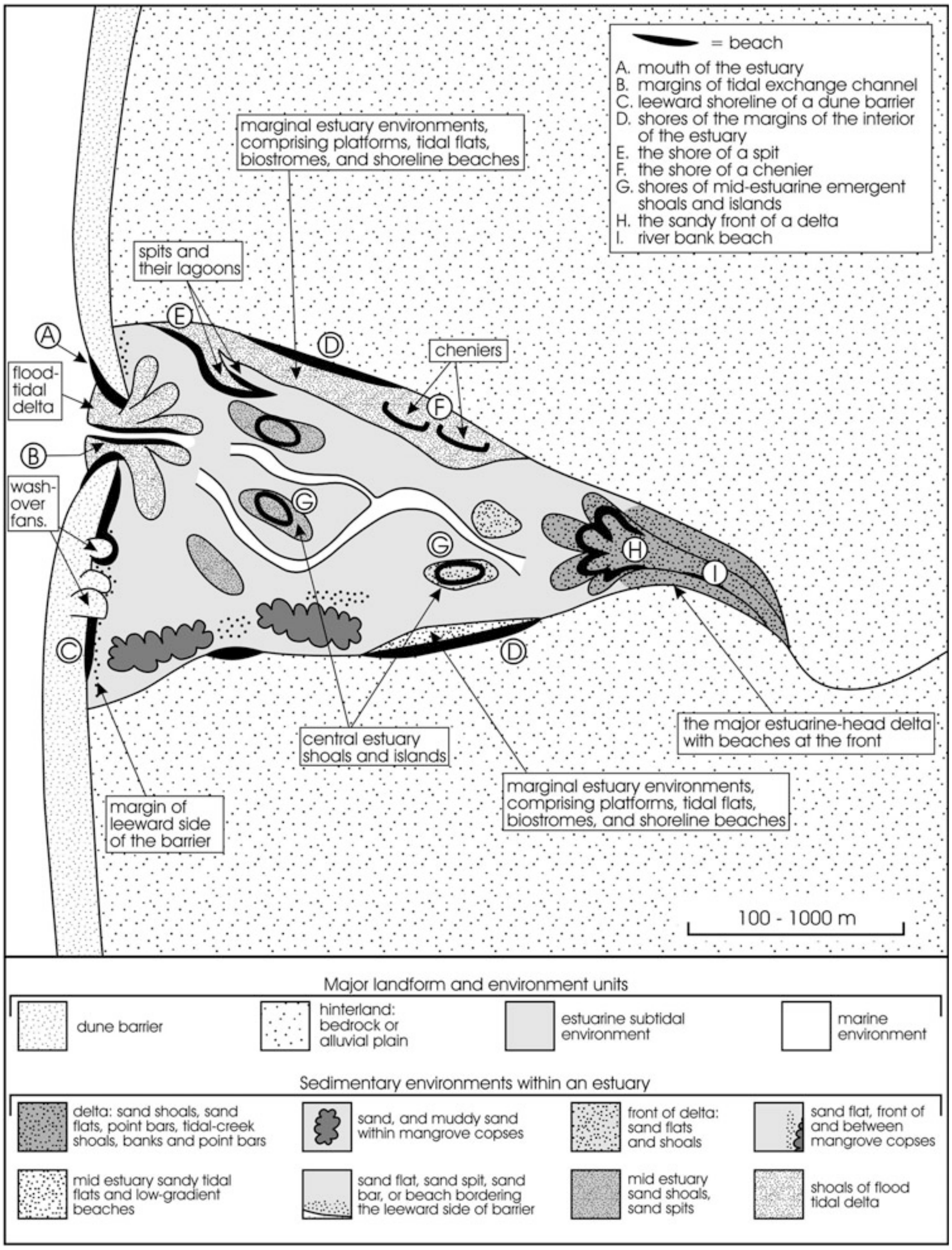
There are a number of locations within an estuary where beaches can form, in general order, from seaward to river; these are (Figure 2; labeled A-I in Figure 2) (1) mouth of the estuary, (2) margins of tidal exchange channel, (3) leeward shoreline of a dune barrier, (4) shores of the margins of the interior of the estuary, (5) shore of a spit, (6) shore of a chenier, (7) shores of mid-estuarine emergent shoals and islands, (8) sandy front of a delta, and (9) sandy sloping bank of a river (a riverbank beach). These shores are locations where sand and/or gravel (that either are preexisting or have been transported to the site by riverine, estuarine, marine, or aeolian processes) is reworked and sculptured by estuarine prevailing wave, storm, tidal, and aeolian processes. These beaches may be small and localized "pocket" beaches varying to large, laterally extensive stretches of shore. The size, slope, and extent of development of a beach in an estuary, and whether it is sandy or gravelly, is a function of a number of factors including the exposure of the shore to wave



Beach Processes, Figure 1 Idealized diagram showing estuaries of the narrow valley-tract type and the wide semi-enclosed type and occurrence of beaches therein. The more complicated, wide, semi-enclosed type of estuary has a larger variety of beaches.

action and storms, the type of sediment that comprises the uplands, the amount of sediment that is delivered to the shore from the estuary or from erosion of the uplands, and the tidal range. A brief description of estuarine beaches in terms of setting, environmental processes, and substrate types is provided later. A selection of beaches in estuaries is shown in Figures 3 and 4.

From seaward to the river along the length of an estuary, the different parts of an estuary vary in relation to the prevailing hydrodynamic conditions. Beaches, for example, are subject to varying oceanic waves, intra-estuarine waves, tides, river currents, floods, and wind, depending on their location. This concept, expressed sedimentologically in facies and stratigraphy along the marine-to-river transition in an estuary, is described by Dalrymple et al. (1992) and is, in principle, also applicable to categorizing and comparing processes and products of estuarine beaches in their locations along, and within, an estuary. Located at the seaward end of an estuary, the hydrodynamic processes are more tide and ocean-wave dominated; located in the central parts of the estuary, the processes are tide influenced, estuarine-wave dominated, and less

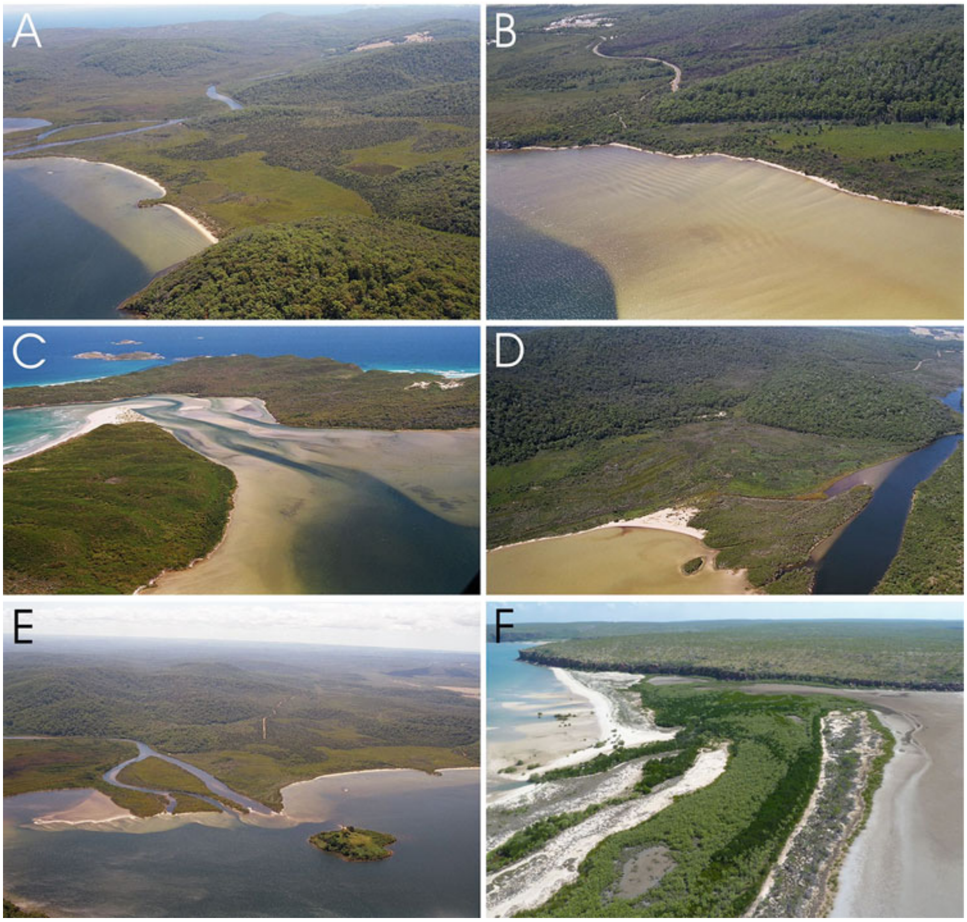


Beach Processes, Figure 2 Using the more complicated, wide, semi-enclosed type of estuary, an idealized annotated diagram showing the various environments of the estuary and where beaches occur. The beaches are labeled A-I.

ocean-wave influenced; and located at the river end, the processes are river-current influenced, estuarine-wave dominated, and less tide influenced.

The shape of an estuary and its relationship to its marine entrance and its relationship to prevailing wind and wind-wave trains also influence the hydrodynamic factors of beaches that affect their shape and sedimentology and the extent that beaches are backed by beach ridges or dunes.

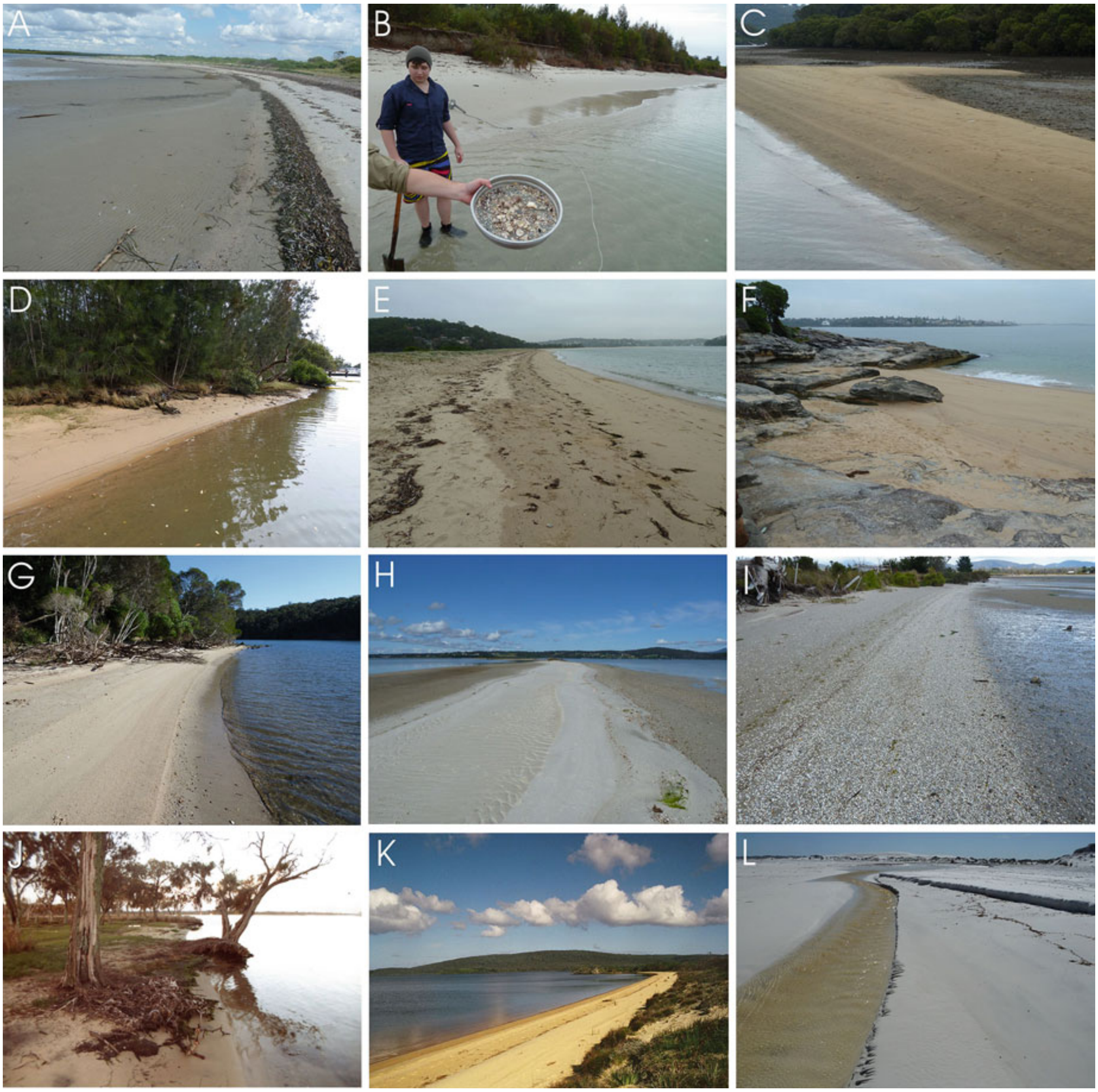
The processes of waves and tides result in different slopes and heights to beaches. Commonly, a beach within an estuary, as a sloping sandy surface, may involve the whole tidal zone of a shore, or only the mid-tidal to high-tidal part of the shore, with the low-tidal part being a low-tidal sand flat, low-tidal muddy sand flat, or low-tidal mud flat. With stronger wave action, beaches within an estuary may have a slope from low- to high-tidal level. Slopes of beaches vary from relatively steep, to moderately sloping, and to low-gradient slopes, and spatially, beaches vary from narrow to wide. Where beaches are exposed to onshore wind, during low tide



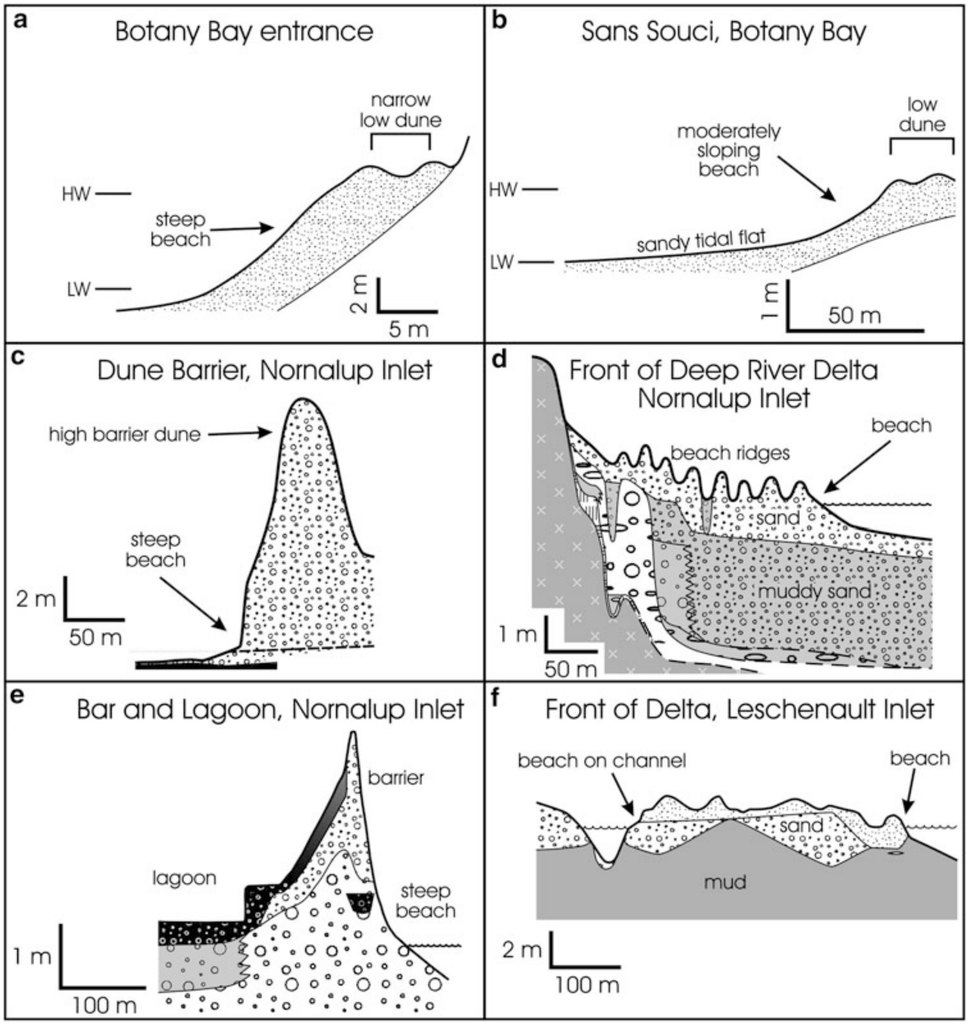
Beach Processes, Figure 3 Aerial photographs of beaches in estuaries in Australia. (**a**) Beach in front of a barrier-and-lagoon complex in Nornalup Inlet estuary, southern Western Australia (Semeniuk et al., 2011). (**b**) Beach in front of a beach ridge system, northern Frankland River Delta, in Nornalup Inlet estuary, southern Western Australia. (**c**) Beaches developed (a) leeward of a dune barrier, (b) at mouth of the estuary, and (c) around shoals in a tidal delta; Nornalup Inlet estuary, southern Western Australia. (**d**) Beach developed at the front of a delta, central Frankland River Delta, in Nornalup Inlet estuary, southern Western Australia. (**e**) Beaches developed along (a) spit at western front of Deep River Delta (to *left*), (b) along sandy central front of the Deep River Delta, and (c) along eastern front of the Deep River Delta (to *right*), in Nornalup Inlet estuary, southern Western Australia. (**f**) Beaches along estuarine-mouth barrier spits, with wave-generated sand ridges, macrotidal Berkeley River Delta estuary, Kimberley Coast, northwestern Australia (Brocx and Semeniuk, 2011); some sand spits and barriers are mangrove vegetated along their lower slope.

(and particularly during low spring tide), aeolian transport sweeps up fine and medium sand from the dry part of the beach to construct a landward low beach ridge above the storm water level. With accretion, the beach ridges may become low dunes. As such, some beaches in estuarine environments are backed by beach ridges and/or dunes. Some profiles of beaches from different estuaries, and ranging from those concavely sloping from high tide to low tide and relatively steep (Botany Bay entrance) to those bordered by a low-tidal flat (Sans Souci, Botany Bay), those bordered by beach ridges/dunes (Deep River Delta, Nornalup Inlet), and those fronting a barrier that shelters a peat-filled lagoon (bar-and-lagoon complex, Nornalup Inlet), are shown in Figure 5.

Wave fields, whether deriving from the open ocean and propagating through the estuary entrance or as wind waves generated within the estuary, in crossing an estuary and impinging on its shores can transport, rework, and winnow sediment to leave a sandy to gravelly deposit, as well as recurringly sorting sediment granulometrically to leave stratigraphically distinctive layers and packages. Storm waves, similarly deriving from open ocean areas or generated within the estuary can transport, rework, and winnow sediment and, additionally, can emplace sediments above the level of the highest tide, often leaving a distinctive stratigraphic, lithologic, and granulometric signature. Combined with wind and wind-induced currents, storms deriving from the ocean can deliver floating debris such as marine algae, seagrass, and shells, as well as chunks of wood, branches, leaves, logs, and the mollusks *Spirula* and cuttlefish skeletons that accumulate on the beach to form a distinctive sedimentary deposit at the



Beach Processes, Figure 4 Examples of beaches in estuaries in Australia showing range of settings, sizes, sediments, geomorphology, slopes, and wrack. (**a**) Narrow sandy moderately sloping mid-tidal to high-tidal microtidal beach with wrack at the *upper* and *lower* limit of slope, bordering a sandy tidal flat; beach is cut into a sandy upland; Botany Bay, Georges River estuary, southeastern Australia. (**b**) Steep sandy microtidal beach extending from low-tidal to high-tidal, and backed by low dune; shells retrieved from the low-tidal part of the beach are shown on the sieve; Botany Bay, Georges River estuary, southeastern Australia. (**c**) Sandy microtidal beach developed on a low-tidal chenier in the mid-tract of the Georges River estuary, southeastern Australia; muddy tidal flats and mangroves occur to leeward of the chenier. (**d**) Microtidal sandy steeply sloping beach developed on the river bank in the mid-tract of the Georges River estuary, southeastern Australia. (**e**) Laterally extensive sandy microtidal low-gradient beach in the central part of the Hacking River estuary (Port Hacking), southeastern Australia; scattered wrack showing high-water levels; beach backed by low dunes. (**f**) Small pocket sandy microtidal beach along the estuarine margin, Hacking River estuary (Port Hacking), southeastern Australia; the beach is interspersed with rocky outcrops. (**g**) Microtidal sandy moderately sloping beach developed along river bank in the estuary of Mallacoota Inlet, southeastern Australia. (**h**) Microtidal sandy beach developed along a spit that emanates from a mid-estuary island in Mallacoota Inlet, southeastern Australia; the beach is bordered by a sandy tidal flat. (**i**) Narrow sandy mid-tidal to high-tidal microtidal low-gradient beach, with lines of scattered wrack, bordering a slightly muddy sand tidal flat; lower part of the beach is cut into a sandy upland; southeastern Tasmania. (**j**) Eroding coast of middle Peel-Harvey Estuary, southwestern Australia (Semeniuk and Semeniuk, 1990) showing cliff cut into peripheral vegetation formations; pocket beaches are developed between the micro-peninsulae. (**k**) Linear extensive microtidal sandy beach, the beach being part of the barrier-and-lagoon system of the northern Nornalup Inlet estuary, southern Western Australia (Semeniuk et al., 2011). (**l**) Exit channel with tannin-stained water at the mouth of an estuary, southern Western Australia; beach shows cliffing.



Beach Processes, Figure 5 Some profiles of beaches in estuaries. (**a**) Microtidal, steep beach backed by rock, at Botany Bay entrance, southeastern Australia. (**b**) Microtidal, moderately sloping beach fronted by wide, sandy tidal flat and backed by low dunes, Sans Souci, Botany Bay, southeastern Australia. (**c**) Narrow, microtidal beach in front of a dune barrier, Nornalup Inlet estuary, southern Western Australia (From Semeniuk et al., 2011). (**d**) Narrow, microtidal, steep beach in the front of beach ridges of the Deep River Delta, Nornalup Inlet estuary, southern Western Australia (From Semeniuk et al., 2011). (**e**) Narrow, microtidal, steep beach in the front of a bar-and-lagoon complex, with a peat-filled lagoon to leeward, Nornalup Inlet estuary, southern Western Australia (From Semeniuk et al., 2011). (**f**) Narrow, microtidal beach in front of the Collie River Delta, Leschenault Inlet estuary (From Semeniuk, 2000).

storm level or high-tide level (Semeniuk and Johnson, 1982; Semeniuk, 1997). Tidal currents perform the same functions of transporting, reworking, and winnowing sediment, creating distinctive lithologic and stratigraphic suites.

Because the provenance of shoreline sediment is variable (reworked from barrier dunes or flood-tidal deltas, and hence marine-derived, reworked as fans, shoestrings, or ribbons from the uplands bordering the estuary, riverine, or generated biogenically), clearly, the sediment type underlying an estuarine beach will vary compositionally and granulometrically (fine sand to coarse sand to gravel and with variable shell content) and, depending on the short-term history of prevailing wave action, storm waves, and tides, will carry a distinctive small-scale stratigraphic signature such as shell gravel lenses interlayered with laminated sand, or coarse sand interlayered with medium/fine sand, or pebble layers in laminated sand, among others.

During a high tide, the beach slope is inundated and the beach sands are saturated. During the ensuing low tide when the beach is subaerially exposed, there are two sub-environments where water resides under a beach: (1) shallow groundwater (that is contiguous with the open waters) whose water table falls and rises with the tide; this water can be referred to as the phreatic zone, where it resides in the pore spaces of the sediments; and (2) water films circumferential to sand particles in the wetted but undersaturated sediment above the water table during low tide; this wet zone can be referred to as the temporary vadose zone and is tidal-flat pellicular water. The surface of the phreatic zone is the groundwater table during low tides. This groundwater table rises and falls with the tides. The phreatic zone determines many of the biological and chemical processes operating under the beach, and the vadose zone during a low tide determines many of the other biological and chemical processes operating under the beach. Depending on the depth to the water table during the period of low tide, the beach may be wet (where the water table is near the surface) or moistened by water rising by capillary action, or may be relatively dry (where the water table is decimeters below the surface).

Location of estuarine beaches

As mentioned earlier, there are nine different environments within an estuary where beaches can form. Therefore, location will determine the suite of processes that combine to form a beach in the first place, the type of sediments that comprise the beaches, and the types of processes that operate postdepositionally on beach sediments.

The most common locations for beaches are the shores of the margins of the interior of an estuary and generally in the central parts of estuaries (i.e., not deltas, spits, cheniers, shoals, and the estuarine mouth). Here, the shores are usually sandy, with sand derived from alongshore, washed up from subtidal zones, reworked from the uplands, or delivered from the marine environment. In microtidal and mesotidal settings, the beach is fronted by low-tidal sand flats. Gravel sources and any eroding rock in estuarine shore environments result in gravelly beaches. Where beaches are bordered by sandy tidal flats and skeletons of shelly benthos are transported onto the beach by waves and storms to form shelly sand, or shell gravel lags, or shell gravel lenses.

The next most common site for beach development is the leeward margin of barrier dunes. Here again, the shores are usually sandy with sand eroded from the dunes, or derived from alongshore, or washed up from subtidal zones. In microtidal and mesotidal settings, such beaches are fronted by low-tidal sand flats which supply shell gravel and shell grit to the beach to form shelly sand, or shell gravel lags, or shell gravel lenses.

The shores of mid-estuarine-emergent shoals and islands also are common sites for the development of beaches. Because shoals and islands present differing aspects to prevailing wind-wave fields, and to wind, there is asymmetry in the suite of processes and in the products developed. Beaches directly facing prevailing waves will have different profiles to those on leeward sides of shoals and islands, and similarly, the sediment response within a beach in terms of lithology, granulometry, and stratigraphic organization will differ from windward side to leeward side of the shoal or island. Beaches peripheral to shoals and sand islands are commonly sandy, while those peripheral to an island of rock can have sandy, gravelly sand, and gravelly beaches.

Beaches can be developed at the mouth of an estuary and along the margins of tidal exchange channels. Beaches at the mouth of an estuary are subject to processes similar to that of open coastal beaches, though the former are more protected; these processes include oceanic wave action, wind-wave action, tides, and wind. These beaches may be backed by low beach ridges built by the prevailing onshore winds. The sediment responses within such beaches in terms of lithology, granulometry, and stratigraphic organization are similar to open coastal beaches and include a larger proportion of floating debris derived from marine sources. Beaches along the margins (banks) of tidal exchange channels are also subject to processes of oceanic wave action, wind-wave action, tides, and wind. Orientation of the channel to the ocean wave field determines how much wave action is involved in shaping the beach morphology and lithology, and, in this context, ebb and flood-tidal currents are more important in that their effects are magnified in (relatively) narrow channels. These shorelines also may be backed by low beach ridges built by the prevailing winds. Beaches at the mouth of estuaries and along the margins of tidal exchange channels are most commonly sandy.

Beaches developed along the shores of spits and cheniers are similar, though these coastal landforms develop in different locations within an estuary, and for spits, there often is a leeward basin. Spits, as linear emergent sandy bars and recurved emergent sandy bars, with one end anchored to a shore, a shoal, or a promontory, are developed along the margins of estuaries in mid-estuarine locations, at the mouths of estuaries, peripheral to shoals and islands, and at the tips of promontories of riverine deltas. Cheniers, as linear emergent sandy bars and recurved emergent sandy bars that are isolated as a sand body, are developed on tidal flats and at the tips of promontories of riverine deltas. The shores of spits and cheniers are developed by prevailing wave action or by storms. Their beach slope is further shaped by tidal currents. For spits, as they are often recurved sand bodies with a leeward lagoon or sheltered area and are subject to hydrodynamic processes on both sides of the sand body, there is a windward beach and leeward beach. Beaches developed along the shores of spits and cheniers are most commonly sandy. For both spits and cheniers, in tropical regions, their sandy leeward (protected) slope often is inhabited by mangroves.

The prograding front of a sandy delta is another location for the development of beaches. In this situation, the beach-constructing agencies are mainly estuarine wind waves, with lesser effect from tidal currents. These beaches are mainly sandy and are peripheral to the delta plain.

The sloping sandy bank of a river is also a location for the development of a beach. Here, the beach-constructing agencies are mainly river current, estuarine wind waves, with lesser effect from tidal currents. These beaches are mainly sandy and locally gravelly. Since these sandy banks are usually protected, in tropical regions, they often support stands of mangroves.

An important aspect of some estuaries is the occurrence of a peat-filled lagoon leeward of a shore-parallel barrier (such as a prograded ribbon of sand or a prograded narrow beach ridge system). With high freshwater levels in the lagoon after rain or floods, freshwater from the peat-filled lagoon may discharge from under the beach slope of the most seaward beach ridge.

Sediment sources

There are four sources of sediment particles that comprise estuarine beaches; these are (1) rivers, (2) estuarine biota, (3) margins of the estuary, (4) preexisting estuarine (relict) estuarine sediment, and (5) marine sources. These sedimentary particles form the "raw material" (viz., the sediment) upon which the physical, biological, and chemical processes in the estuary in the beach environment will act and imprint various structures and products.

River sediment, sediment from the margins of an estuary, and marine sediment are exogenic, deriving from outside the estuarine basin and being delivered to the estuary. Estuarine biota generates endogenic sediment.

Rivers deliver sand, some gravel, and mud, with sand and mud being the dominant particle types. The margins of the estuary through erosion by waves, tidal currents, wind, and (rain) sheetwash provide sand, mud, and gravel. Depending on the type of upland, i.e., whether it is rocky, sandy, or muddy, the sediment eroded into the margin of the estuary by waves, tides, rain, and storms varies from coarse, medium, and fine sand to gravel, mud, or mixtures of these. Preexisting estuarine (relict) sediment is sediment within the estuary relict from an earlier depositional phase that is remobilized by waves, tides, and storms and transported onto the beach. Though it may comprise a range of grain sizes such as sand, mud, and (rock and/or shell) gravel, only sand, rock, gravel, and shell are of relevance to estuarine beaches. Delivery of marine sediment to the estuary (and specifically to estuarine beaches) can be complicated and occurs in three pathways: (1) through the entrance or mouth of the estuary by tidal currents, wind-induced currents, and waves; this sediment is coarse, medium-to-fine sand, rock gravel, and shell; and this source also includes the floating mollusks such as *Spirula* and cuttlefish; (2) initially accumulating from the marine environment as a dune barrier which, on its leeward margin, is then eroded by estuarine processes of waves, tides, and sheetwash into the estuarine shore zone; this sediment usually is medium and fine sand; and (3) aeolian transport from the dune barrier into the estuarine environment; this sediment is usually fine sand.

Endogenic biogenic sediment in an estuary is variable depending on biogeography, climate, and salinity of the environment. At macrofaunal scale, it ranges from molluskan shell and their fragments (deriving from benthic infaunal bivalves, epifaunal gastropods, and oysters that colonize rocks or form biostromes, mussels that also colonize rocks or form biostromes), crustacean fragments, ostracods, and minor components such as bryozoans, echinoderms, hydrozoans, and sponges. Shelly fauna that contributes shell to the sediment in an estuary varies in composition according to their location in terms of salinity and the variability of salinity (Day, 1981): (1) near the estuary mouth where the environment is marine, organisms are stenohaline marine in character; (2) in mid-estuarine environments, biota is truly estuarine or euryhaline marine in composition (the latter are species represented in the marine environment but which tolerate salinities of 5–50 ‰); (3) in the river mouth and river channel where it is freshwater, biota is euryhaline freshwater (i.e., species primarily are freshwater types, but some tolerate salinities greater than 5 ‰) or stenohaline freshwater. At microbiota scale, biota that contributes to estuarine sediment includes foraminifera and diatoms. Some of these biotas inhabit beach environments and so directly contribute to skeletal accumulation in situ, but a majority live externally to the beach and are transported to the beach site.

Seagrass; algal beds; algal mats; saltmarsh; mangroves logs, stumps, and branches; mangrove leaves; and terrestrial wood and logs washed into the estuary from rivers also contribute to beach sediment. Seagrass and algal beds such as *Heterozostera*, *Halophila*, *Ruppia*, *Zostera*, *Chaetomorpha*, *Fucus*, *Gracilaria*, and *Ulva* occur in the intertidal or shallow subtidal zone and are delivered to beaches by waves, tidal currents, and storms. Leaves of saltmarsh species and mangroves and wood are washed onto the beach from alongshore (and, for saltmarsh, also from behind the beach), particularly during storms. Although not growing on the beach, estuarine plants from these various intertidal and subtidal environments are transported from their habitat to accumulate on the beach as "flotsam and jetsam" (also termed "wrack"), as scattered plant fragments or as organic beds (incipient peat) of plant matter some 10–50 cm thick. Most typically, plant material from their respective habitats occurs as scattered fragments and detritus on the beach slope to be buried later in the beach sediment.

The physical, biological, and chemical processes operating on an estuarine beach

Processes operating on a beach can be categorized as physical, biological, or chemical. Many of the processes on beaches are ubiquitous within beaches throughout an estuary, but there is a distinct suite of processes in different parts of the estuary, depending on the extent of wave action, tidal currents and river currents, wind, the nearshore or low-tidal biota (and what is delivered to the beach as shell and plant matter), the type of macrobiota resident under and on the beach, the type of microbiota resident under and on the beach, the amount of sediment sheetwash delivered onto the beach from the adjoining upland, the amount of freshwater seepage onto the beach face, salinity of the estuarine waters, the chemistry of the groundwater and pore waters under the beach (dependent on estuarine setting), and the hydrological (groundwater) through-flow.

These physical processes, biological processes, and chemical processes and their products are described below. Which process(es) on or under the beach is/are dominant is determined by where the beach is located in the estuary, how active are the hydrodynamic processes, the extent of mobility of the sediments (and therefore to what extent the organic matter is turned over), the amount of influx of organic matter, and the extent of oxidation of the sediments.

Sandy beaches commonly exhibit gradients normal to their shore, e.g., a gradient in inundation and evaporation, with attendant gradients in wave energy and tidal energy, and hence a graded expression of the processes of sedimentation, erosion, and hydrochemical effects (Brocx and Semeniuk, 2009). This results in variable, complex, and diverse physical, biological, and geochemical products across the shore and variation in fine- to small-scale stratigraphic sequences.

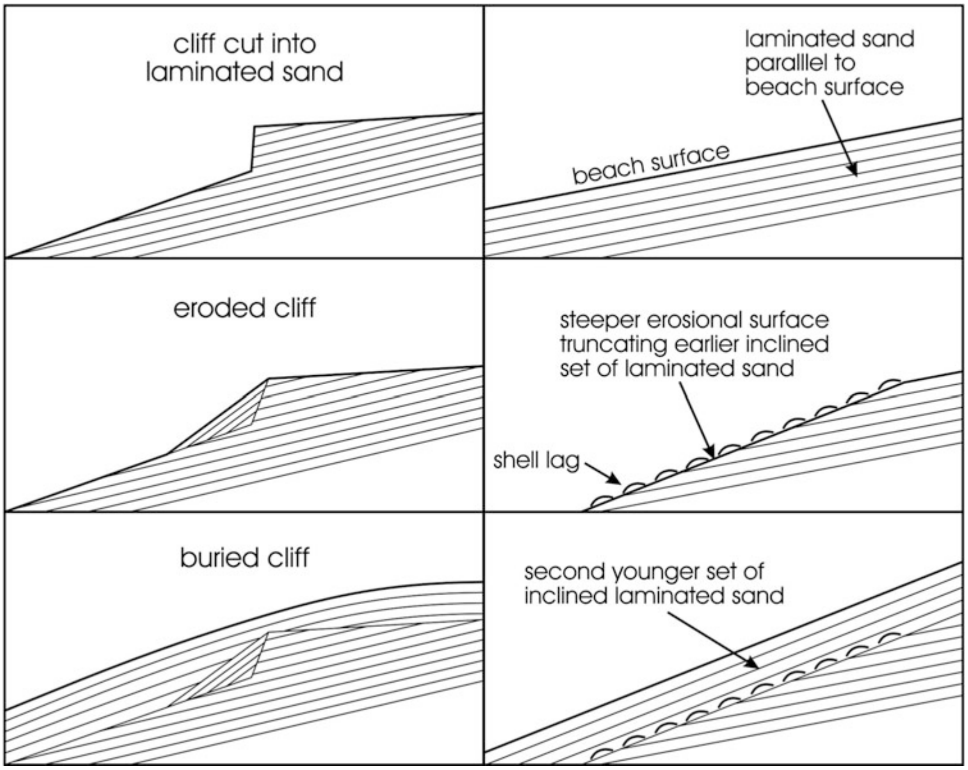
The physical processes on beaches are wave action by prevailing wind waves and by oceanic waves propagating through the estuary mouth, wind-generated currents, tidal currents with maximum currents during spring tides and lower-velocity currents during neap tides, formation of cliffs and cusps, wind activity acting on dry beach surfaces and on wet beach surface, storms resulting in chaotic wave trains and waves often with elevated water levels, evaporation, freshwater seepage into the beach and freshwater upwelling from under the beach, gas upwelling from under the beach, rainfall effects (such as rain infiltration), and wave-swash infiltration.

Wave action, tidal currents, and storms are involved in sedimentation processes to develop beach sand lamination. Wave action, tidal currents, and storms transport sediment, and, depending on the wave energy, tidal-current velocity, and degree of storm activity, they sort and separate sediments into grain-sized suites of sand with grains of similar specific gravity (a monomineralic sand, siliciclastic sand of quartz and feldspar grains, or siliciclastic sand and carbonate sand) or hydraulically equivalent suites (e.g., fine sand-sized grains of magnetite as spheres with specific gravity of 5.2 are hydraulically equivalent to medium sand-sized quartz and feldspar grains as spheres with specific gravity of 2.6, and 2.6–2.7, respectively; Tourtelout, 1968; Selley, 2000). Wave action, tidal currents, and storms also transport and sort shells into size-graded and oriented accumulations (Behrens and Watson, 1969; Reineck and Singh, 1980).

With run-up and backflow during wave action on a beach slope, sediments, once sorted, are deposited as granulometrically distinct and/or compositionally distinct laminae that, with accretion, result in laminated beach sand with laminae alternating in grain sizes, grain-sized suites, or in composition (e.g., quartz fine sand laminae alternating with quartz medium sand laminae, or with mixed quartz fine to medium sand laminae, or with grain-thick micro-laminae of rutile very fine sand or silt). Wave action and tidal currents, during the high tide when the beach slope is inundated, winnow the sand of the beach slope leaving a lamination-scale lag of quartz medium sand and coarse sand and laminae of opaque (heavy) minerals such as rutile, tourmaline, and magnetite. Where there is shell, or shell fragments, the action of waves, tides, and storm waves can concentrate these particles leaving laminae of shell, shell fragments, and shell grit within the sand laminae. As such, with accretion, the beach is underlain by laminated sand, with lamination defined by grain-sized variation, shell layers, shell grit and fragments, and laminae of opaque minerals (heavy minerals). Lamination that is formed by waves, tides, and storms under the beach slope is parallel to the sloping surface so that, with beach-slope accretion, the laminations of the sand in the beach environment are inclined towards the estuary.

With a change of season and change in wave dynamics, or in the change from spring tide to neap tide, or with storms where water levels are higher than normal and wave action is intense, chaotic, and short-term repetitive, the beach slope (with erosion or accretion) can change its inclination. Where such erosion is followed by accretion, the erosional surface is marked as a horizon of truncation of the underlying inclined lamination and accretion of the additional laminated sediment takes place parallel to the horizon of truncation. These horizons of truncation are preserved as bedding discontinuities in the small-scale stratigraphic record. Where there are small channels or basins eroded into the sloping beach (scour is effected by wave run-off, formation of beach cusps, tidal drainage run-off, low-tidal seepage from the beach slope, and freshwater seepage channels), followed by filling and accretion of these channels and scours by later laminated sand, there is the development of small-scale (10–50 cm wide and 5–20 cm deep) cut-and-fill structures.

During storms, or periods of intense and sustained wave action that may be atypical of prevailing conditions, or during the change in water level from spring to neap tide, or a change in the wave climate inter-seasonally, the beach may erode to form a steeper beach slope or to form a cliff (Figure 6). For such beaches, the steeper beach slope is reflected in a change in dip of layering and lamination with a pre-erosion set of lamination less inclined than the post-erosion set of lamination. The interface between the sets of lamination can be marked by a lag deposit of shells or pebbles (Figure 6). Where a cliff has



Beach Processes, Figure 6 Cross sections of beaches showing macroscopic internal structures produced by erosion followed by accretion with the beach changing its slope, and internal structure, where a cliff is cut into a beach, is cliffed, and then later buried by accretion.

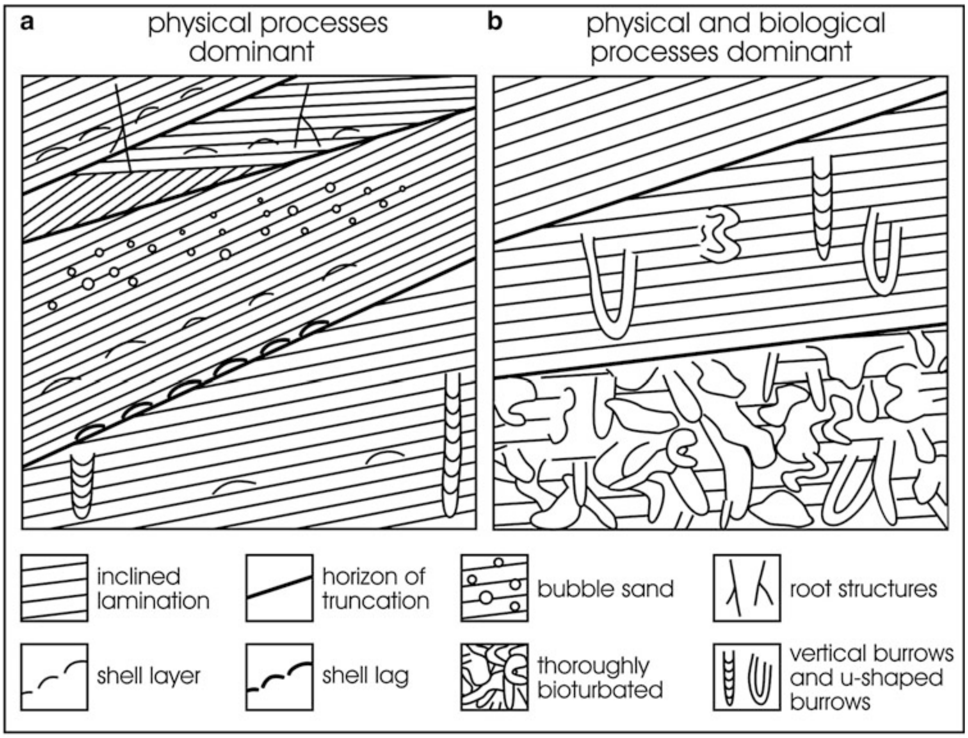
formed, it may be eroded to a more gentle slope and buried such that it is marked by a prominent discontinuity in the stratigraphic profile.

Beach cusps or rhythmic shoreline features are regular triangular, temporary-constructed accumulations of sand and/or gravel projecting from the shore (Komar, 1976). The positive cusps (or horns) alternate with depressions (or embayments). Generally, they are a few to several meters in size and spacing. Usually, the cusps are comprised of sediment that is of coarser materials than that comprising the adjoining embayments, e.g., the cusps may be comprised of coarse sand or shell gravel, while the embayment is comprised of medium sand. Cusps and their embayments manifest a stratigraphically diagnostic internal geometry and sedimentary structures (Reineck and Singh, 1980). If there is a marked grain-sized difference between the horns, this is also evident stratigraphically. The origin of beach cusps is still being debated (Rasch et al., 1993). Originally, it was thought that standing edge waves (waves perpendicular to the shore) interacting with incoming wave trains created the conditions for development of a cusp-and-embayment morphology (Guza and Inman, 1975; Guza and Bowen, 1981). However, Werner and Fink (1993) and Coco et al. (2000) provide an alternative model, i.e., the self-organization theory, wherein feedback processes between currents and sediment response result in a self-organized pattern to develop cusp and embayment on a regular spacing.

The wind has several effects on the beach. It can transport fine sand and medium sand from the dry parts of the beach leaving a lag of coarser grain sizes. In the extreme, the deflation of the beach and removal of finer grain sizes leaves a lag of coarse sand or of shell and shell fragments that form an “armored” surface of platy grains on the beach (van der Wal, 1998). Wind also dries the sediment, ripples the sediment, and constructs adhesion ripples.

The beach surface, particularly if low-gradient, may be rippled. Ripples are formed subaqueously by low-energy wave action and by tidal currents. Ripples are formed subaerially by wind on dry parts of the beach. On wet beaches, subject to strong wind where the wind is delivering dry sand from elsewhere, adhesion ripples are formed (Reineck and Singh, 1980). Adhesion ripples are oriented, linear accumulations of sand that adhere to the wet surface by surface tension and microscopically accrete forming small sand ribbons internally comprised of undulating convex-upward laminae.

With the rising and falling of the tide and concomitant rising and falling of the water table of the phreatic zone under the beach, together with the swash run-up, air is



Beach Processes, Figure 7 (**a**) Structures produced in beach sediment where physical processes are dominant. (**b**) Structures produced in beach sediment where physical processes and biological processes are co-dominant.

forced out of the aerated zone but trapped by descending swash water. As such, air is trapped in bubbles in the sand in the upper tidal level. In areas with low wave action but large tidal range (mesotidal and macrotidal), with a rapidly rising water table, air is also trapped in the sand to form air bubbles. Where air is entrapped in the beach sand, the structure is termed "bubble sand" and is a distinctive structure of sand in tidal zone (Emery, 1945; De Boer, 1979; Reineck and Singh, 1980). In estuaries, it occurs in all beaches with a tidal fluctuation (viz., mouth of the estuary and margins of tidal exchange channels, leeward shoreline of a dune barrier, beaches along the margins of the interior of the estuary, the shores of spits and cheniers, and of mid-estuarine emergent shoals and islands); it is less developed to absent on beaches that comprise the sandy front of deltas.

As noted earlier, storms and wave action during times of elevated water levels are also instrumental in developing lithologically distinct sedimentary deposits (Semeniuk and Johnson, 1982; Semeniuk, 1997). These may be marked by the concentrated occurrence of marine and estuarine plant wrack, wood and log debris, shell deposits, and accumulations of floating mollusks such as *Spirula* and cuttlefish. The marine-derived accumulations of floating mollusks are more common on beaches near estuarine mouths or within estuaries that are widemouthed and have a strong marine influence at their seaward parts.

A summary of the products of the physical processes acting on beach sediments is shown in Figure 7a.

Freshwater through-flow from the uplands bordering an estuarine beach, or from water ponded by a beach barrier, can discharge over or through a beach. With a beach barrier that bars/ponds a freshwater lagoon to leeward, or where the uplands provide general sheet flow of freshwater, the seepage across and through the beach can be a broad front (a seepage front, or interface). Such seepage may not be perennial but linked to the wettest time of the year. On the other hand, due to drainage channels and buried drainage channels from the uplands, or because of hydrological conduits, the freshwater through-flow may be channeled and restricted in its passage in corridors across and in the subsurface through the beach. The through-flow of freshwater across and through a beach will have biological, hydrochemical, and geochemical effects (see later). In particular, it may affect the composition of macrobiota and microbiota that are ecologically linked to a specific salinity.

Freshwater discharging under a beach, because of its buoyancy in relation to denser marine or brackish water, can escape to the beach surface in a discharge "pipe." This water escape, or freshwater upwelling from under the beach, results in physical disruption of the lamination of beach sediment. The upflow can entrain sand and bring it to the surface. The lamination within and in an aureole around the discharge "pipe" is contorted, and the surface of discharge is often marked by a small sand mound ("sand volcano") some 30–50 cm in diameter and up to 10 cm in height.

Methane, hydrogen sulfide, and ammonia gases generated by decomposition of organic matter buried under the beach can upwell along a preferred conduit, escaping to the surface of the beach. Such gas upwelling also causes physical disruption of lamination of beach sediment.

Evaporation, caused by solar radiation or by wind, results in the loss of moisture. Depending on the depth to the water table under a beach, evaporation can induce an increase in salinity by moisture loss and in precipitation of salt (halite).

During rainfall, meteoric water effects three processes: dilution of surface water and pellicular water salinity, dissolution of any halite that has precipitated on the surface, and vadose water-induced infiltration. For the latter, rain washing onto a beach during its exposure at low tide can deliver dust or any fine-grained sediment on the beach to levels lower down the sediment profile. If the beach is not reworked later by waves and tides, this material can be preserved as meniscus sediment. Waves and tides washing over a beach slope on a rising tide can also infiltrate the beach sand vertically and deliver fine-grained sediment (that was in suspension in the water) into the beach-sand pore spaces.

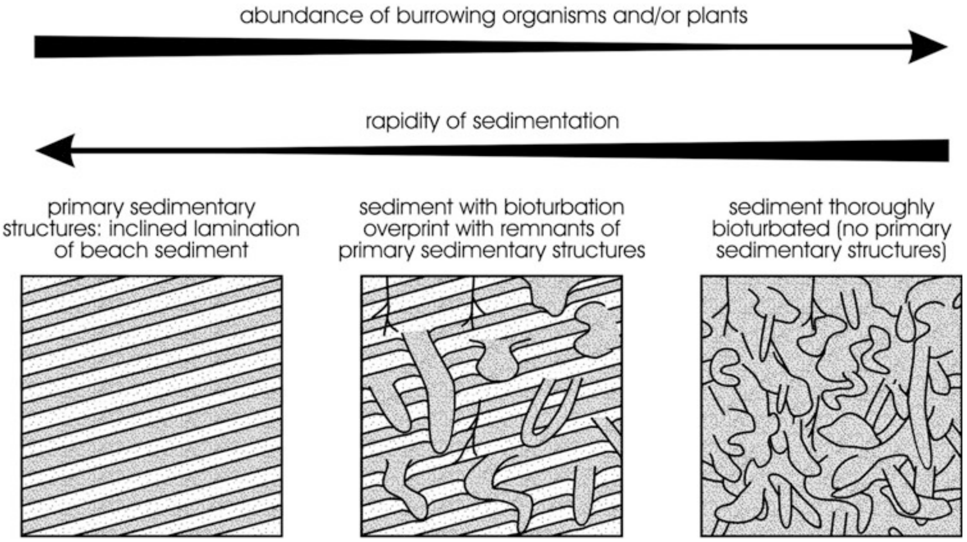
In terms of the horizontal sequence of small-scale landforms, sedimentary structures, and processes, sandy beaches provide excellent examples of the products of wave and tidal energy intersecting a sloping shore and illustrate the range of sedimentary products that are developed across the slope gradient from shallow subtidal to supratidal, in response to the graded effect of waves, tides, wind, and freshwater seepage (Clifton, 1969; Clifton et al., 1971; Reineck and Singh, 1980; Semeniuk and Johnson, 1982; Semeniuk, 1997; Brocx and Semeniuk, 2009). For instance, wave action intersecting a sloping shore is translated from a lower flow regime (varying progressively upslope) to an upper flow regime, and the resultant upslope progressive development of rippled beds and perhaps megarippled bedforms further upslope, and plane beds. Hourly, daily, weekly, and seasonal variation in wave patterns, coupled with storm effects, tide fluctuation, and onshore winds, generate lamination, shell layers, cut-and-fill structures, discontinuities (Mii, 1958), variation in grain size across laminations, and bubble sand. While the literature cited above on beach processes and products is derived mainly from beaches on oceanic shores, the principles of sedimentation and stratigraphic evolution apply equally to estuarine beaches. Moreover, the beaches closer to the estuary mouth, particularly in wide valley-tract, ocean-facing estuaries, have many features in common with oceanic beaches.

The biological processes on beaches are macrobiota shell production; microbiota test production; burrow construction; root structuring; general bioturbation; fragmentation; macrofaunal and meiofaunal breakdown of seagrass, algae, and other plant materials on the beach face; microbial decomposition; sediment pigmentation by sulfides; and hydrochemical changes in pH, Eh, and ionic chemistry effected by microbiota. The conspicuous products of biological activity result in shell layers, burrows, bioturbation, and pigmentation of sediments.

Shell production results in articulated bivalve shells being preserved in situ in the sediment (e.g., pipis and tellinids) or, where shells are disarticulated and locally transported after death, in shells being scattered in the sediment parallel to lamination usually in a convex-up orientation, though concave-up orientations are possible (Nagle, 1967; Reineck and Singh, 1980; Savarese, 1994). Gastropods are often predators of bivalves in the shore environment and are responsible for their death (the evidence being drill holes in the disarticulated bivalves; Carriker and van Zandt, 1972; Kabat, 1990), after which follows disarticulation. Gastropods also scavenge for decaying organic material on beaches. Gastropods contribute shell to beach sediments after their own death. Often bivalve and gastropod shells form laminae of shell concentrates in the beach sediment, with the bivalve shells specifically also forming a platy shell pavement on the surface due to wind deflation or current winnowing. Microbiota, such as foraminifera and diatoms, contribute tests as fine sand-sized particles that accumulate as fine-grained calcareous and siliceous particulates in the sediment.

Some of the major products of biogenic processes on beaches include burrow construction, general bioturbation, and root-structuring (McCall and Tevesz, 1982). Fauna that live in permanent burrows on the beach slope create distinct biogenic structures. Bivalves and beach worms are examples of such fauna (Reineck and Singh, 1980; Brown and McLachlan, 1990). The burrows may be diffuse, vertical structures penetrating the beach lamination (formed as the animal migrated vertically in response to changing groundwater levels), or may be a simple single tube open burrow that the animal has lined with organic matter or mud to prevent collapse, or may be a u-shaped open burrow. With the open burrows, remaining open because of their lining, later sand infiltration into abandoned burrows brought in by wave swash or tidal currents results in a sand-filled tube that is penetrative through the beach lamination. Bioturbation of the sediment is produced by other animals that burrow in the beach sediment but do not produce permanent tube dwellings.

Depending on whether physical processes that produce beach lamination are dominant over biological processes that produce burrow and bioturbation structures, the beach sediments can grade from laminated sand and shelly sand with occasional burrows (i.e., physical processes are dominant), to laminated sand and shelly sand with abundant burrows and bioturbation structures but within which the beach lamination is relict and still evident, to thoroughly bioturbated sand and shelly sand with some sand-filled vertical burrows evident (i.e., biological processes are dominant). Figure 8 illustrates this gradation in laminated sediment to bioturbated sediment.



Beach Processes, Figure 8 The progressive obliteration of primary sedimentary structures in beach sediments reflecting the relative balance between biota abundance and the rapidity of sedimentation. The primary sedimentary structures, once diagnostic of an environment, are reduced to root-structured or burrow-structured sediments and then finally to a thoroughly bioturbated sediment.

Beaches, of course, are commonly shaped and internally structured by physical processes, but macroscopic biological processes can become important enough to co-dominate in the development of the beach sediment structure. Figure 7b illustrates structures produced in beach sediment where physical processes and biological processes are co-dominant.

Burrow structures and bioturbation affect sediment macrobiologically, microbiologically, hydrologically, and geochemically through aeration, providing conduits for micro-hydrological through-flow, altering composition of meiofauna, nutrient recycling (such as nitrogen fluxes), and diagenetic mineral overturning (e.g., pyrite in deeper anoxic sediment oxidized at the sediment surface), among other processes (McCall and Tevesz, 1982; Alongi, 1985; Aller, 1988; Dittmann, 1993; Sadao, 2002; Webb and Eyre, 2004).

The higher parts of a beach, storm levels of a beach, and/or the low beach ridge immediately leeward of the beach slope often are colonized by halophytes and other strand vegetation. Such plants result in root structuring of sediments.

Animal predation, bioturbation, and foraging result in shell fragmentation and shell comminution. Crabs, fish, stingrays, octopus, and shorebirds hunt and feed on various invertebrate fauna of sandy shores, resulting in the invertebrate exoskeleton fragmentation. Animal bioturbation and sediment ingestion also results in shell fragmentation.

After storms, or after some active wave action that might disrupt the seagrass beds and algal beds in the nearby subtidal environments, the plants living on nearby rocky shores, or the saltmarsh and mangroves from high-tidal environments, plant material is transported to the sandy shore and the beach may be littered at the high-tide mark by varying plant debris (flotsam and jetsam, or wrack). A range of macrofauna, such as crabs, or gastropods, or avifauna, forages among this material digesting it, or feeding on the organisms that inhabit it (Griffiths et al., 1983; McLachlan, 1985; Dugan et al., 2003; Lewis et al., 2007). This results in the breakdown of the plant material on the beach. Smaller organisms specialized for this wrack environment, such as amphipods, isopods, and meiofauna, also consume the finer-grained plant material, adding to the biological breakdown of plant and other organic matter stranded on the beach slope (Hayes, 1974; Poulin and Latham, 2002; Mews et al., 2006; Pelletier et al., 2011). In addition, particularly on wet beaches, there is microbial decomposition of organic matter on the beach and of organic matter shallowly buried at or below the water table of the beach (Jørgensen, 1982; Lovley and Phillips, 1986; Henriksen and Kemp, 1988).

Organic matter on and under beaches can also be broken down microbially (decayed). Some of this microbial decay is related to, fixed on, or mediated by structures and larger organisms in the environment (e.g., where microbes, meiofauna, and bacterial productivity are associated with tubes constructed by a polychaete; Alongi, 1985). Microbial decay involves the conversion of organic molecules to inorganic molecules and ions. This biotransformation is often subsumed under the term “mineralization.” It is the process by which organic matter is “mineralized” (transformed to inorganic compounds, radicals, or elements) by fermenting, denitrifying, sulfate-reducing, and methane-producing bacteria (Jørgensen, 1982), some under anaerobic conditions and some under aerobic conditions. In estuarine beach environments, this takes place below an anaerobic water table or in the aerated vadose zone of the sandy beach. One major pathway of microbial decay, for instance, involves the breakdown of organic molecules and their oxidation by sulfate-reducing bacteria, which utilize the sulfate exogenically in the environment as the energy source for the decomposition, and in the process, organic matter is removed from the sandy beach environment.

Bacterial reduction of sulfate to sulfide is responsible for the oxidation of organic matter buried in sediments (Lovley and Phillips, 1986; Machel, 2001) that concomitantly results in pigmentation of light-colored sediments to grey or black. The sulfate ion is common in seawater, sediment, or in waters rich with decaying organic material, and sulfate-reducing bacteria are common in anaerobic environments where they utilize the sulfate ion as an electron donor, aiding in the degradation of organic materials. Sulfate reduction is the dominant terminal step in the biomediated mineralization processes of sulfate-rich sediments where the sulfate reducers inhibit the methanogens by competing for common substrates. This sulfate reduction is quantitatively important in the overall oxidation of organic matter (Barton and Fauque, 2009).

Various minerals can be precipitated by biomediation, the best known being iron sulfide (pyrite) and calcite. If Mn, Cu, and Zn are present in the environment, they also can produce sulfides. Generally, Fe is the most common transition metal cation in natural environments, so Fe sulfide (as pyrite) is the most common mineral. Precipitates of pyrite are commonly framboidal (framboids being small clusters of pyrite crystals resembling a raspberry <1 μm in size, but ranging from 0.5 to 40 μm in size; Wilkin et al., 1996; Sawlowlicz, 1993; Schieber, 2002). While organic-matter-rich sediments inherently tend to be black or dark grey, the fine-grained precipitated iron sulfide disseminated throughout the sediment as a result of bacterial decay of organic material similarly renders sediments to various shades of grey to black. Calcite can be precipitated in association with microbial activity, particularly in wet parts of a beach (e.g., that associated with cyanobacterial mats; Kremer et al., 2008).

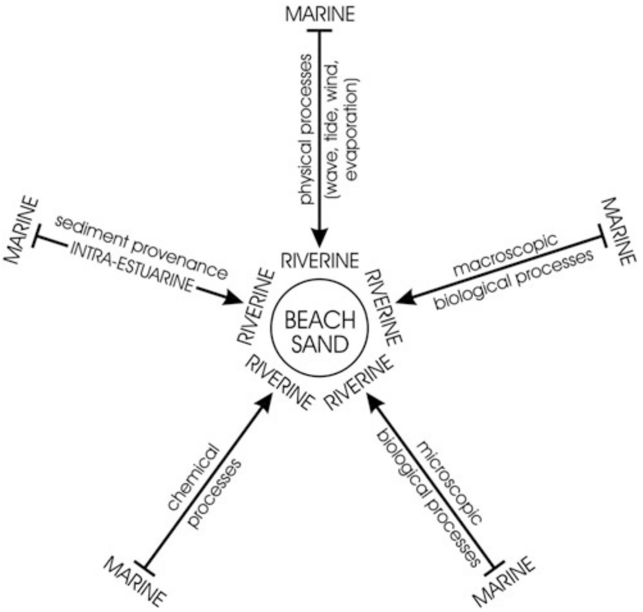
Microbial changes in the sediments leading to pyrite precipitation and sediment pigmentation carry with them hydrochemical changes in pH and redox conditions (i.e., Eh). Groundwaters under beaches often are anoxic and with the biomediated transformation taking place that result in the formation of sulfides; the groundwaters can become acidic and markedly oxygen depleted. From a generally alkaline state for seawater, the pH may decrease to 6.5 or 6.0 in the pore water of beach sediments. The Eh may be negative, with any decrease in Eh generally being related to the decrease in the dissolved oxygen in pore waters (Zobell, 1946; Haraguchi, 2012). Weakly acidic groundwater dissolves shell that is buried in beach sand such that shells lose their luster, appear corroded, or may be completely dissolved away. (In this context, the microbial activities that have been subsumed under "biological processes" grade into "chemical processes" in that the acidity of the groundwaters has increased as a result of biological activity, and this affects carbonate mineral solubility).

Ionic chemistry is also affected by microbial activity. The extent that sulfate and phosphate are microbially utilized in the environment, determining the depletion of sulfate ion and affecting phosphate concentrations, is an example of how biological (microbiological) processes affect ionic concentrations (e.g., Jansson, 1987; Lovley, 1991) and, vice versa, how the resulting ionic chemistry dictates development of precipitates (Berner and Raiswell, 1984). Such microbial activity also changes sulfide concentrations and, with precipitation of sulfide, changes the (transition) metal concentration in waters of species such as Fe and Mn.

In terms of chemical processes, estuaries in general, with their variety of environments ranging from deltas, shallow water sand platforms, tidal mud flats and sand flats, saltmarsh and mangroves, subaqueous shoals to deep water mud beds, among others, manifest a diversity of chemical processes and a variety of authigenic and diagenetic minerals, particularly where there is interaction between microbiota, anoxic sediments, muddy sediments, and different hydrochemical fields (cf., Cook, 1973; Cook and Mayo, 1980; Pye, 1984; Pye et al., 1990; Rasmussen et al., 1998; Hedges and Keil, 1999; Pirrie et al., 2000; Aller, 2004; Bush et al., 2004; Michalopoulos and Aller, 2004; Byrne et al., 2011). Sandy beaches and gravelly beaches, with their well-drained and more aerobic conditions and limited grain composition, however, represent the relatively low diversity end of the spectrum of chemical processes that occur in estuaries and present a more limited range of possible chemical processes and products.

The chemical processes on estuarine beaches are dissolution, precipitation of minerals (authigenesis), biomediated mineral precipitation, diagenesis of minerals, diagenetic structure development, sediment pigmentation (e.g., pyrite mottling), the effects of freshwater throughflow, and the oxidation of organic matter. The products of precipitation, often resulting in color mottling, in cemented laminae, or in development of nodules, commonly occur as diagenetic overprints on a primary sediment (i.e., either laminated, burrowed, or thoroughly bioturbated).

Note that in the context of precipitation of minerals, the concepts of authigenetic minerals and diagenetic minerals can overlap. The broad definition of authigenesis is of a mineral generated in situ. These would include mineral precipitates deposited on the estuary floor. At the mineral level for the process of mineral precipitation and/or alteration, the broad definition of diagenesis is the mineralogical alteration of one mineral to another. At the larger scale, e.g., at the sediment level where minerals are crystallizing in the sediment pore spaces lithifying the sediment, the cementing agents are considered by many authors to be diagenetic, but since they are crystallizing in situ, they are also considered by other authors to be authigenic. In this contribution, minerals precipitated from estuarine waters, regardless of whether they are open



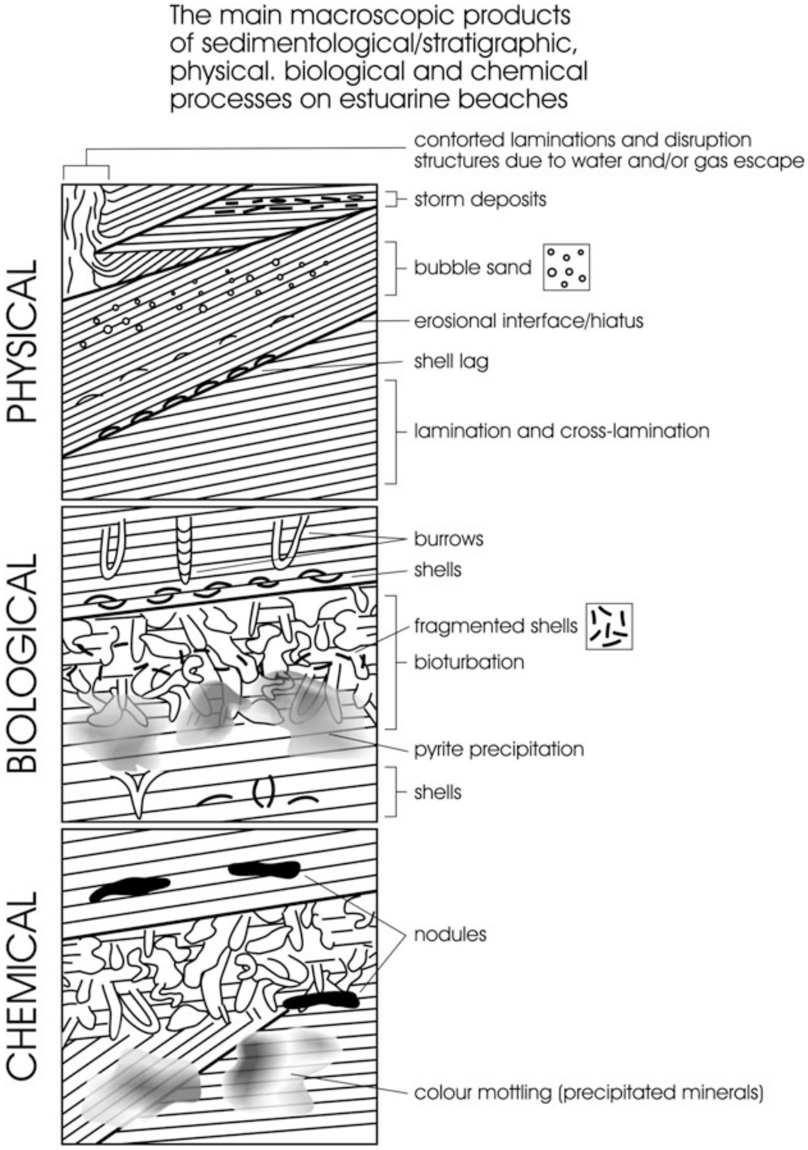
Beach Processes, Figure 9 The core of the diagram shows beach sediment. The influences and imprints on this beach sediment to generate variety in the lithology, structures, and products from physical, chemical, and biological processes are illustrated along five separate axes: (1) the provenance of the beach sediment with origin from river sources, marine sources, and intra-estuarine that influences primary lithology; (2) the gradient of physical processes operating on the beach (i.e., the gradient of hydrodynamic and aerodynamic conditions), grading from marine dominated near the estuary mouth to river dominated at/near the river mouth, to the effects of wind; (3) macro-biological processes, such as shell production, burrowing and bioturbation, environment-diagnostic shell assemblages, and shell fragmentation, grading from marine-dominated biotic effects near the estuary mouth to river-dominated biotic effects at/near the river mouth; (4) microbiological processes, such as biomediated mineralization, decay, and pyrite formation, grading from marine-dominated effects near the estuary mouth to river-dominated effects at/near the river mouth; and (5) chemical processes, such as solution, mineral precipitation, and diagenesis, specific to sites that are marine dominated near the estuary mouth grading to river dominated at/near the river mouth.

estuarine waters or intra-sediment pore waters, are authigenic minerals, and those formed by alteration of preexisting authigenic minerals or of sand grains are diagenetic minerals.

Dissolution of carbonates and precipitation of pyrite are the main chemical process on and under beaches. Dissolution of carbonates is a feature of chemical processes in estuaries (Abril et al., 2003). Under beaches, with acidic groundwaters, carbonate shells (composed of calcite, Mg calcite, or aragonite, or mixtures of these minerals) are corroded. They pass through various stages of corrosion (from lustrous shell, to shells lacking luster, to pitted shells) to ultimately dissolve away. The changing chemistry of the phreatic zone and vadose zone during high and low tides can result in the precipitation of minerals. In highly evaporative climates, with evaporation effected by solar radiation and/or winds, the surface of the beach wetted with saline pellicular water can evaporate to precipitate halite which forms a surface crust, termed salcrete (Yasso, 1966). Under a beach, particularly in tropical estuaries, depending on ionic concentrations, minerals such as aragonite, Mg calcite, and calcite may precipitate interstitially and cement the sand to varying degrees (cf., Bathurst, 1975), though carbonates can also precipitate in beach sand of temperate climates (Arrieta et al., 2011). In environments with alternating pH and with an appropriate Fe content and Eh in the groundwaters, Fe minerals can precipitate (Boyle et al., 1977). Authigenic minerals in freshwater environments in beaches under deltas and headwaters of estuaries include iron minerals and carbonate minerals, while those under beaches in marine salinities towards the ocean part of the estuary can be carbonate minerals. Mineral precipitation is manifest in beach sediments as lithification, color changes, mottling, or nodule development. Some of the biogeochemistry of estuaries and their sediments are described by Bianchi (2007).

Authigenic mineral precipitation can result from organic matter decomposition (Berner, 1981), with the mineral species related to sedimentary setting and location in the estuary. The main minerals precipitated are carbonates, sulfides, phosphates, and amorphous silica (Suess, 1979). The precipitation of minerals in the beach sand can result in the local development of diagenetic structures such as color mottling due to pyrite or to iron oxides, thin ferricrete sheets, ferricrete nodules, and carbonate nodules.

Freshwater through-flow on a beach changes the groundwater salinity from the prevailing marine or brackish salinity to lower salinity concentrations. This affects macrobiota



Beach Processes, Figure 10 Environment-specific processes and products as preserved geohistorically in the evolving stratigraphy under the beach. Three sections are diagrammatically illustrated: lithology and stratigraphy produced by physical processes, lithology and stratigraphy produced by biological processes, and lithology, stratigraphy, and overprints produced by chemical processes.

assemblages that are infaunal under the beach, the composition of microbiota, and some of the geochemical interactions. As described earlier, freshwater through-flow can occur along a broad interface or can be channelized. If freshwater seepage is in a broad front, its chemical effects will be along the interface of beach and upland, and will be shore-parallel. If channelized, the effects will be in specific locations along the beach. If the freshwater derives from upslope peat beds (as described earlier), the seepage will be more acidic than prevailing beach groundwaters and will result in dissolution of the more susceptible shells, alteration of macrofauna composition, and alteration of microbiota composition. These alteration effects will be along a broad front along the upper part of the beach or, if seepage is channelized, in local patches.

Freshwater may also flow over the beach and, by this process, the freshwater affects the hydrochemistry and geochemistry at the sites of entry onto the beach and sites of infiltration into the beach sand. Under the beach, organic matter can vary in content from scattered detritus to peat beds but, under aerial conditions and/or through-flow of freshwater, can oxidize. In time, in such situations, organic matter in upper parts of the beach is depleted by oxidation.

Summary

What may be viewed as a relatively simple system, the beach, underlain mainly by sand, can in fact frequently manifest a variable and spatially complex system. In an estuary, the beach, as a shoreline deposit, spans the range of environments from the river entrance to the marine estuarine mouth. Estuarine beaches, whether as a long continuous shoreline or as a discontinuous set of pocket beaches, traverse three major environments in terms of hydrodynamic setting, hydrochemistry, macroscopic biological setting, microscopic biological setting, and sediment provenance. As such, the beach in estuaries is subject to five major environmental gradients (Figure 9).

In terms of hydrodynamic setting, there is the part of the estuary located at/near the marine environment that is dominated by ocean waves and, to a lesser extent, by intra-estuarine wind waves, tides, and onshore winds; there is the central estuary dominated by intra-estuarine wind waves, wind, and lesser effects from tides, river current, and floods. There is the riverine part that is dominated by river currents, wind waves, wind, and, to a lesser extent, tides. In terms of hydrochemistry, there is the marine part that is dominated by marine salinities and the attendant effects on biota and their biological processes and marine authigenesis/diagenesis. There is the central estuary dominated by fluctuating salinities or brackish waters, the attendant effects on biota and their biological processes, and estuarine authigenesis/diagenesis. There is the riverine part that is dominated by freshwater and its attendant biological and authigenesis/diagenesis products. In terms of biological setting, there is the marine part that is dominated by marine assemblages. There is the central estuary dominated by euryhaline biota specialized for estuarine conditions. There is the riverine part that is dominated by freshwater biota. In terms of sediment provenance, the tripartite subdivision of estuaries is reflected in the exogenic sedimentary particles (those derived outside of the estuarine basin) in that there is a marine component dominantly towards the estuarine mouth, a mixed component in the central estuary, and a riverine component towards the river mouth. The tripartite subdivision of estuaries also is reflected in the composition of endogenic sedimentary particles and sediment types (those derived inside the estuarine basin) in that peat and bioclasts (shells) are diagnostic of freshwater parts of the estuary (though peat is not within the beach environment; its presence leeward of beaches adds hydrochemical complexity to freshwater seepages). As such, the beaches of the estuary provide a framework to viewing and studying beach processes across the longitudinal range of estuarine environmental variability.

This variability of beach setting within the estuary and the beach processes relative to beach setting is expressed geomorphologically, stratigraphically, lithologically, biologically, and authigenically/diagenetically.

At smaller scales, the physical, biological, and chemical processes operating on beaches result in environment-specific features such as sedimentary structures, specific suites of lithology such as laminated sand, or concentrations of shell and rock gravel, shell lenses, burrow structures, bioturbation, and chemical products. These environment-specific processes and products are preserved geohistorically in the evolving stratigraphy under the estuarine beach (Figure 10).

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Cross-references

Evaporation and Transpiration
Mineralization
Stratigraphy of Estuaries
Tidal Flat Salinity Gradient

BENTHIC ECOLOGY

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Definition

Benthic ecology is a subdiscipline of ecology that focuses on organisms living in or on the bottom of a water body (e.g., an estuary) and the interactions among these organisms and with their surrounding environment.

Expanded definition

E. P. Odum (1971) defined ecology as "the science of interrelations between living organisms and their environment." The word "benthic" is derived from "benthos" defined as the bottom of a water body and/or the organisms living on the bottom of the water body (Websters II New Riverside University Dictionary, 1994). Thus, benthic ecology encompasses the study of the interrelations among organisms living in or on the bottom of a water body (e.g., an estuary) and their interactions with the surrounding environment. Benthic organisms include megafauna (>>>1 mm) such as bottom-oriented fish, crustaceans, and echinoderms living at or just above the sediment surface; macrofauna (>0.5 or 1 mm) such as polychaetes, molluscs, anemones, and arthropods living on top of or within the sediment; meiofauna (0.1 mm to 0.5 or 1 mm) such as nematodes, oligochaetes, and harpacticoid copepods living in sediment interstices (spaces between grains of sediment); and microfauna (<0.1 mm) such as protozoans (Miller, 2004; Levinton, 2009). Benthic organisms also include benthic diatoms, attached algae, kelp, and seagrass, as well as the associated bottom microbial community. In addition to biological and community interactions, benthic ecology includes chemical transformation and physical modifications of the environment as mediated by the benthos and the effect of these transformations and modifications on associated ecological communities (Levinton, 2009; Day et al., 2012). For example, benthic organisms can influence nutrient cycling and hydrodynamics through their activities (e.g., bioturbation, reef building, seagrass bed expansion), while hydrodynamics, depth, and other environmental factors can act to structure benthic communities. Benthic ecology examines a wide variety of organisms and habitats from the intertidal to the deepest bottom of the ocean. The science of benthic ecosystems is as diverse and interconnected as the seafloor itself.

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Cross-references

Biogenic Sedimentary Structures
Infauna
Macrofauna
Meiofauna
Microphytobenthos
Nutrient Dynamics
Oyster Reef
Soft Sediment Communities

BIOACCUMULATION

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Definition

The accumulation of contaminants, pollutants, and/or their metabolites into animal or plant tissues along a period of time which typically represents the degree of exposure of the individual to the chemical element, species, or compound present in the environment where it lives.

Fundamentals

Contaminants, pollutants, and their environmental metabolites (chemical agents) can be found in all the different biogeochemical compartments (air, water, soil/sediment). These agents can be both organic and inorganic. The accumulation of a chemical agent by living organisms depends on the fraction of it that is chemically and physically available to the biota. The chemical form of contaminants and pollutants present in the environment will define the pathway, higher or lower uptake of the chemical agent by the organisms, and consequently its bioaccumulation. Bioaccumulation is therefore a natural phenomenon that becomes even more relevant when the chemical element or compound in question is distributed in the environment in concentrations above its natural level or, in the case of synthetic compounds, is present even in minimum amounts (Clark, 2001).

Contaminants and pollutants are present in the air, soil/sediments, and water of every environment on the surface of the Earth, where they arrive via direct release or short- to long-range transportation. The biota is then inevitably exposed to environmental contaminants and pollutants released by every economic and social activity known. All biological groups present will, in theory, be exposed (Chen et al., 2012; Melwani et al., 2013). However, their susceptibilities vary according to taxonomic group and ecological function. This exposure means that there will be (in)direct contact of the chemical with the individuals and therefore biochemical interaction between them. For an element or chemical compound to be bioaccumulated, it must be first incorporated via one of the biological processes of respiration, feeding/digestion, or skin absorption (Clark, 2001). Through respiration, contaminated water or air enters in contact with tissues specialized in efficient gaseous exchanges (i.e., gills or lungs). This facilitates the passage of the contaminant through cell membranes and vascular walls, from which it gains the circulatory system and is distributed throughout the body. The most common (and efficient) way for an aquatic animal or plant to assimilate and accumulate elements and compounds in their tissues is via feeding and digestion of contaminated food sources (Chen et al., 2012; Melwani et al., 2013). If food is contaminated with toxic/harmful chemicals, it can, during digestion, release then in the digestive tract. Therefore, the pollutant is absorbed through the intestine walls, together with nutrients, and also falls into the circulatory system to be distributed. Accumulation of such elements and chemicals occurs preferentially in the different tissues of plants and animals. Some tissues have functions, structures, and compositions more prone to the accumulation of different elements and compounds. Some examples are the liver, kidney, brain, and fat tissues.

Bioaccumulation is, to a certain extent, reversible. If exposure ceases, metabolism can eventually excrete the accumulated chemical back to the environment. Bioaccumulation is a biological phenomenon related to each individual and can be examined at tissue level when necessary. It is worthy of note that the bioaccumulation concept refers to the tendency of a certain chemical agent to be accumulated by the biota through all sources of ambient, i.e., by water and food. Bioaccumulation differs from bioconcentration since bioconcentration refers to the tendency of a certain chemical agent to be accumulated by biota only from the water.

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Cross-references

Bioavailability
Biomagnification

BIOAVAILABILITY

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Definition

An ecological property presented by chemical elements and compounds in the environment (chemical agents) that determines if they will be more, or less, efficiently assimilated by the biota that enters in contact (is exposed to) with them.

Fundamentals

The bioavailability of a chemical corresponds to the amount of the element that can be absorbed by the living organisms from the environment (Chen et al., 2012). It is a parameter directly associated with the chemical species of this element present in each biogeochemical compartment (Hoffman et al., 2012; Sinoir et al., 2012). This characteristic of chemical agents in the environment can then be time and space dependent as water quality changes along ecological gradients and seasons. Bioavailability is a descriptive property of chemical elements, chemical species, and compounds determined by a relatively complex group of factors, indicating their own chemical characteristics, the chemical and physical characteristics of the medium they are distributed in, metabolism rates, and the type of exposure the biota has (skin, breathing, feeding). Bioavailability can increase or decrease according to the combination of these factors (Chen et al., 2012; Hoffman et al., 2012; Sinoir et al., 2012). The same chemical can have its bioavailability change with the presence and concentration of Cl^- ions, as in seawater, for example, or due to changes in water temperature, organic particulate loads, or dissolved oxygen. In the same way, bioavailability varies if an element changes its oxidative state (e.g., Cr^{+3} vs. Cr^{+6}). Chemical agents must be bioavailable in the environment in order to be assimilated, bioaccumulated, and possibly biomagnified in the biotic compartment. Usually, once a chemical agent enters the trophic web, it becomes readily available for all its subsequent levels. Contaminated food is a common form of bioavailability (Chen et al., 2012).

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Cross-references

Bioaccumulation
Biomagnification

BIOCHEMICAL OXYGEN DEMAND

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Synonyms

Biological oxygen demand

Definition

Biochemical oxygen demand (BOD) is the amount of molecular oxygen required to oxidize organic matter into a stable inorganic form through aerobic microbial decomposition.

Description

The biochemical oxygen demand (BOD) is determined by empirical testing in which standardized laboratory procedures yield the relative oxygen requirements of wastewaters, effluents, and polluted waters (APHA, 1999). Five days at 20 °C is often used to oxidize the carbonaceous organic matter, being referred to as "$BOD_{5,\ 20}$."

Importance: This test is important for pollution control. Heterotrophic microbial metabolism transforms biodegradable organic compounds into stable or mineralized end products, including water, carbon dioxide, sulfates, phosphates, ammonia, and nitrates. The BOD test is widely used to assess the level of domestic or industrial sewage pollution discharged in estuaries. Dissolved oxygen (DO) consumption by bacteria during organic matter regeneration is an indirect indicator of estuarine water quality.

Impacts: This process can consume dissolved oxygen (DO) faster than the atmosphere can supply it through diffusion or the autotrophic community (algae, cyanobacteria, and macrophytes) can produce it. Decomposition of organic matter may fully deplete oxygen from the water (Kennish, 1997). Since less dissolved oxygen is available in the water, fishes and other aquatic organisms may not survive.

Analytical Method: This method of determination is based on dissolved oxygen (DO) measurements. In the first measurement, two or more bottles of water samples are collected. The oxygen is measured in sample 1 on the first day, and 5 days later, it is measured in sample 2. Next, the BOD is calculated by subtracting the results. The BOD may reach 7 mg/l in productive estuaries compared with values higher than 7 mg/l in polluted estuaries (APHA, 1999).

Estuarine Dynamics: Aquatic plant photosynthesis raises the DO during the day, and respiration lowers it at night in estuaries (Day et al., 2013). This leads to a large diurnal variation in the availability of dissolved oxygen. Meteorological variations and estuarine dynamics have a large influence on the dilution and transport of organic matter. During high tide, more oxygenated coastal waters are encountered, increasing the availability of dissolved oxygen. The lower dissolved oxygen levels are generally found in lower-salinity regions in the upper estuary. When the runoff is high, more freshwater enters the estuary often transporting higher loads of organic wastes into the system.

Limitation: BOD measures the pollution potential. It is an indirect quantification of the potential impact, not a direct measurement of such impact. $BOD_{5,\ 20}$ does not detect nonbiodegradable matter. It does not consider toxicity, nor does it inhibit effects from materials on microbial activity because it only measures the oxygen consumed in a standardized test. The BOD is a subset of the chemical oxygen demand (COD).

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Cross-references

Dissolved Oxygen
Eutrophication
Heterotrophic
Nonpoint Source Pollution
Oxygen Depletion

BIOGENIC SEDIMENTARY STRUCTURES

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Synonyms

Ichnofossils; Trace fossils

Definition

Biogenic sedimentary structures are evidence of organism–substrate interactions preserved in rocks and sediments, including those recorded in estuarine environments. Their study is termed "paleoichnology" (from the Greek *palaios* = old, ancient and *ichnos* = a trace, a track), whereas similar studies in modern sediments are referred to as "neoichnology." Markings that do not reflect the behavior of organisms (e.g., marks made by the shells of dead mollusks passively transported on the seafloor by waves and/or currents) are excluded from the trace fossils. In addition, biogenic sedimentary structures do not include body fossils (direct remains, such as shells, bones, teeth, etc.) or molds of organism bodies.

Introduction

Organisms that have adopted endobenthic or epibenthic modes of life produce biogenic sedimentary structures by "disturbing" the substrate. The number of biogenic sedimentary structures is vast, and various authors have proposed subdividing them into component groups to better define their significance (e.g., Frey, 1971, 1973; Frey and Pemberton, 1984; Pemberton et al., 1992; Bromley, 1996). Four major categories of structures produced by the activities of organisms are generally accepted:

- Bioturbation structures, which reflect the disruption by organisms of biogenic and physical stratification features or sediment fabrics, include tracks, trails, burrows, and similar structures.
- Biostratification structures, which consist of stratification features imparted by organism activities, include certain stromatolites, biogenic graded bedding, byssal mats, and similar elements.
- Biodepositional structures, which reflect the production or concentration of sediments, include coprolites, fecal pellets, pseudofeces, and fecal castings.
- Bioerosion structures, which are mechanically or biochemically produced by organisms in rigid substrates, include borings, rasps and scrapes, bites, drill holes, and related traces.

These categories, and others proposed in the literature, are not exhaustive because the divisions among the various categories are vague. For example, plant–arthropod interactions may be revealed by biogenic structures preserved in wood, leaves, and seeds, which are not strictly rigid substrates comparable to rockgrounds or hardgrounds.

Consequently, the appropriate placement of this group in one category or another is unclear. Egg cases are not usually described as trace fossils, but eggs can be preserved within a fossil nest, providing direct evidence of reproductive behavior. In that sense, they fall within the realm of paleoichnology and are often placed under "other evidence of activity."

The conceptual framework

The importance of paleoichnology in traditional fields such as paleontology, paleoecology, sedimentology, and stratigraphy derives from the peculiarities of trace fossils, which reflect both their mode of formation and their taphonomic histories. Unfortunately, the limitations of trace fossil also arise from these basic characteristics ("ichnological principles" of Bromley and Fürsich 1980; Ekdale et al., 1984; Bromley, 1996; Pemberton et al., 2001). The examples are as follows: (1) A long stratigraphic range can limit the use of trace fossils in biostratigraphy. (2) A narrow environmental range may reflect similar responses of tracemakers to a given set of paleoecological parameters, and therefore, biogenic sedimentary structures tend to occur preferentially in certain depositional environments. The combination of (1) and (2) greatly facilitates the comparison of rocks of different ages formed in similar depositional settings. (3) The rarity of secondary displacement means that trace fossils are very rarely transported and therefore represent the original environmental position of the tracemakers (i.e., they are in situ fossils). This characteristic reveals the strength of ichnofossils in paleoecological reconstruction. (4) Non-preservable soft-bodied trace producers must be considered since many biogenic sedimentary structures record the activities of soft-bodied organisms that are usually not preserved because they lack hard parts. This fact highlights once again the difference between trace and body fossils. (5) Peculiar occurrences in otherwise nonfossiliferous sediments are very often the result of diagenetic processes that, on the one hand, enhance the potential preservation of trace fossils and, on the other, may obliterate the tests and shells of body fossils. (6) The same individual or species of organism may produce different structures corresponding to different behavior patterns; this characteristic can produce compound traces, where intergradational forms reflect the transition from one behavior to another. (7) The same individual may produce different biogenic structures, reflecting the same behavior on different substrates; this peculiarity is attributable to variability in the substrate conditions in terms of the degree of consistency, grain size, and stratal position. (8) Conversely, identical (or very similar) structures can be produced by systematically different organisms, where their behavior is similar; this peculiarity makes it impossible to establish a one-to-one relationship between tracemakers and biogenic structures. (9) A single structure may reflect the activity of two or more organisms, living together or in successive times, within the substrate (the "composite" traces of Pickerill, 1994). Paleoenvironmental research based on these characteristics represents the majority of contemporary ichnological studies and applications.

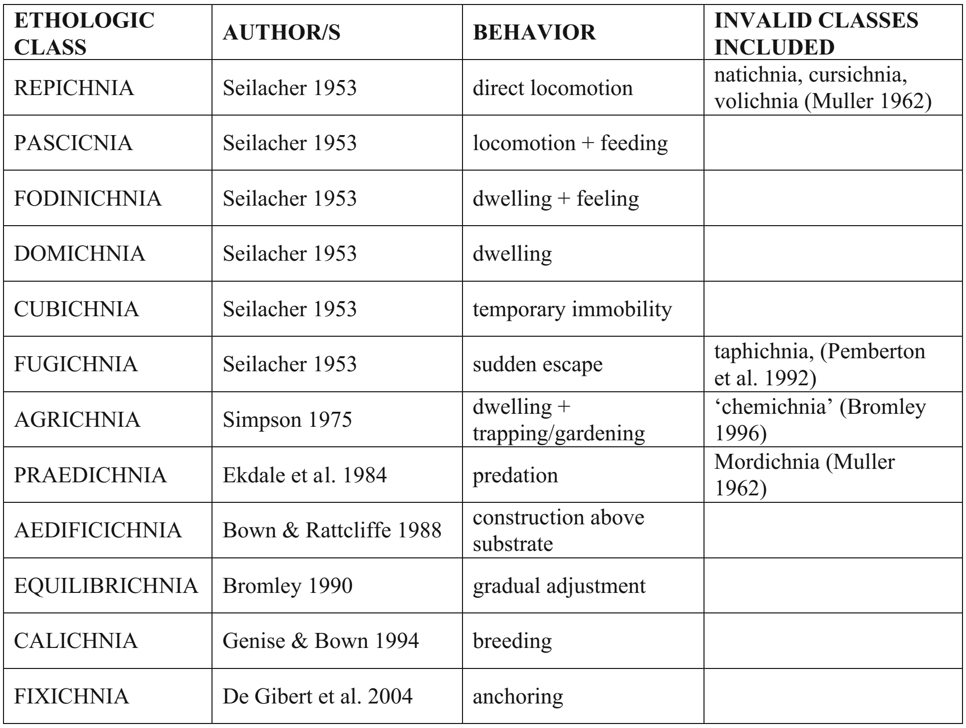
Naming biogenic sedimentary structures

The use a formal taxonomy by ichnologists must accommodate the many difficulties that arise from both the historical background and the intrinsic nature of ichnofossils. In the early years of paleoichnology, a large number of invertebrate trace fossils were named and described as the remains of algae or other organisms (Age of Fucoids by Osgood, 1975). However, based on the priority law, many of these names are taxonomically valid, such as *Cruziana*, *Zoophycos*, and *Chondrites* erected as algae and *Nereites* as worms.

The 1964 edition of the International Code of Zoological Nomenclature (ICZN) ruled that trace fossil names erected after 1930 were to be accompanied by a statement on the identification of the tracemakers. Because fulfilling that requirement is essentially impossible, all post-1930 trace fossil names (ichnotaxa) were formally unavailable, whereas the pre-1930 taxa retained their valid names but were treated on the same basis as body fossils. This is considered the beginning of the Dark Age of Ichnotaxonomy (Bromley, 1996). Thanks to the long-lasting and determined activities of ichnologists and exhaustive scientific debate, ichnofossils have finally been bounded by the ICZN in 1985. The 4th edition of the ICZN (1999) includes in the "work of animals" all trace fossils. This means that animal, protistan, plant, and fungal trace fossils are considered in exactly the same way as zoological taxa in terms of the availability and validity of their names. However, they are called "ichnotaxa" ("ichnogenera" and "ichnospecies") to distinguish them clearly from true biotaxa. The significant departures with respect to body fossils (see also the previous section) further complicate trace fossil taxonomy. For example, according to the ICZN, only fossil specimens should be named, which prevents ichnologists erecting ichnotaxa based on recent biogenic structures that might be assigned very often to their producers on a case-by-case basis. Under these circumstances, some authors prefer to name the tracemaker associated with the recent structure, whereas others opt to use the prefix "incipient" before the ichnotaxon (e.g., incipient *Thalassinoides*) (Bromley and Fürsich, 1980). A separate code for naming trace fossils, as proposed by Sarjeant and Kennedy (1973), might be a possible alternative to circumvent the aforementioned difficulties, but this prospect has never gained legal standing.

Classification of trace fossils

Although the recent ICZN explicitly encompasses ichnofamilies, there is no true ichnotaxonomic superstructure above the rank of ichnogenus, and trace fossils can be grouped together in several ways. Traditionally, the most



ETHOLOGIC CLASS	AUTHOR/S	BEHAVIOR	INVALID CLASSES INCLUDED
REPICHNIA	Seilacher 1953	direct locomotion	natichnia, cursichnia, volichnia (Muller 1962)
PASCICNIA	Seilacher 1953	locomotion + feeding	
FODINICHNIA	Seilacher 1953	dwelling + feeling	
DOMICHNIA	Seilacher 1953	dwelling	
CUBICHNIA	Seilacher 1953	temporary immobility	
FUGICHNIA	Seilacher 1953	sudden escape	taphichnia, (Pemberton et al. 1992)
AGRICHNIA	Simpson 1975	dwelling + trapping/gardening	‘chemichnia’ (Bromley 1996)
PRAEDICHNIA	Ekdale et al. 1984	predation	Mordichnia (Muller 1962)
AEDIFICICHNIA	Bown & Rattcliffe 1988	construction above substrate	
EQUILIBRICHNIA	Bromley 1990	gradual adjustment	
CALICHNIA	Genise & Bown 1994	breeding	
FIXICHNIA	De Gibert et al. 2004	anchoring	

Biogenic Sedimentary Structures, Figure 1 List of acceptable ethological classes according to De Gibert et al. (2004) (Modified).

important classifications include preservational, phylogenetic, and behavioral schemes, although virtually all classifications are to some extent genetic because they presuppose that the structures were produced biogenically.

The preservational aspect takes into account two main facets: (1) the physiochemical processes of preservation and alteration and (2) the toponomy (or stratinomy). The former facet falls within the realm of diagenesis, which is of paramount importance in trace fossil preservation; nevertheless, no classification based on diagenetic features is yet available. The latter focuses on the description and classification of biogenic structures in terms of their mode of preservation and occurrence. Toponomic schemes have been devised by various authors (e.g., Simpson, 1957; Seilacher, 1964; Martinsson, 1970), and most of these relate to the position of a trace fossil to the main casting medium. The schemes of Martinsson (1970) and Seilacher (1964) have a lot in common and have gained the greatest acceptance.

Phylogenetic classification attempts to establish a correspondence between a trace fossil and the potential producer, a fascinating target but very difficult to reach. This is because ichnofossils usually reflect animal behavior and reflect their anatomy or morphology to a much smaller extent. As stated in the previous section, a single taxon may construct different biogenic structures, and conversely, identical (or very similar) structures may be made by different taxa. It is sometimes possible to match tracemaker and trace fossil, but this problem must be approached with caution, bearing in mind that generalizations should be avoided and each occurrence of a given ichnofossil must be treated on an individual basis.

Above all, trace fossils are good indicators of the behavior of animals, and it is therefore not surprising that ethological classification has been extremely successful. The original scheme proposed by Seilacher (1953), based on five categories, has been progressively modified and enlarged by various authors; among them are Frey (1973), Ekdale et al. (1984), Ekdale (1985), and Bromley (1996). Frey and Pemberton (1985) suggested that categories be restricted in number and that new proposals are only justified if they are well founded on new behaviors. Today, a dozen categories are generally accepted (Figure 1), although it must be emphasized that the overlap among groupings is unavoidable, reflecting the intergradation inherent in nature.

Ichnofacies model

According to the concept proposed by Seilacher (1964, 1967), ichnofacies are trace fossil assemblages that recur through long intervals of time and are typical of a given set of environmental conditions (Frey and Pemberton, 1985). Ichnofacies are named after a characteristic ichnogenus and may be recognized even if the namesake form is absent. The classic marine ichnofacies, those named for *Nereites*, *Zoophycos*, *Cruziana*, and *Skolithos* by Seilacher (1967), were originally based on the fact that many of the parameters controlling the distributions of the tracemakers tend to change progressively with increasing depth. Because these bathymetrical relationships are potentially very valuable for paleoenvironmental reconstruction, the ichnofacies sequence has long been regarded as a relative paleobathymeter. Today, it is well known that ichnofacies are essential for the reconstruction of depositional settings, but paleobathymetry constitutes only one aspect because the distribution of tracemakers is controlled by a number of interrelated ecological/sedimentological parameters, including the sedimentation rate, substrate grain size, salinity, oxygen level, turbidity, light, temperature, and water energy (Pemberton et al., 1992). Because these parameters may occur at specific water depths, it should not be surprising to find nearshore assemblages in offshore sediments, and vice versa. For example, the *Skolithos* ichnofacies, which is typical of nearshore settings, may occur in offshore tempestites or deep-marine turbidites, and the *Cruziana* ichnofacies, which is typical of lower shoreface to offshore deposits, may also be present in shallower settings, such as intertidal flats on tide-influenced shorelines (Miller, 2007).

In recent decades, ichnologists have proposed many new ichnofacies from continental and marine environments, some of which are considered well founded, some are retained as mutually equivalent, and still others are considered invalid categories (see Buatois and Mangano, 2011 for a detailed discussion). In a recent paper, Knaust and Bromley (2012) recognized 14 formally defined ichnofacies among those that conform to Seilacher's paradigm. Five of them encompass the marine to marginal-marine softground substrates: *Psilonichnus*, *Skolithos*, *Cruziana*, *Zoophycos*, and *Nereites*. Three are regarded as substrate-controlled (omission) ichnofacies and are very useful for delineating surfaces, with sequence-stratigraphic implications: *Glossifungites*, *Trypanites*, and *Teredolites*. Six ichnofacies encompass the continental realm: *Scoyenia*, *Mermia*, *Coprinisphaera*, *Termitichnus*, *Celliforma*, and *Octopodichnus–Entradichnus*.

Ichnology and estuarine systems

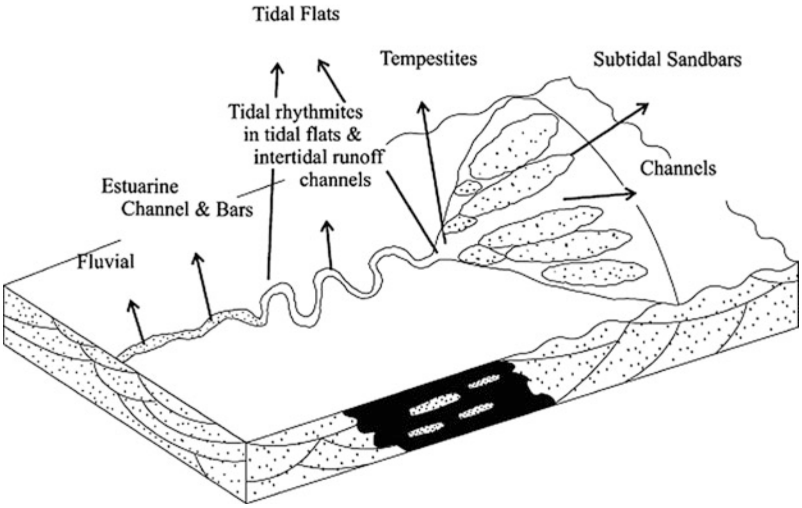
According to Dalrymple et al. (1992), an estuary is "the seaward portion of a drowned valley system which receives sediments from both fluvial and marine sources and which contains facies influenced by tide, wave, and fluvial processes. The estuary is considered to extend from the landward limit of the tidal facies at its head to the seaward limit of the coastal facies at its head." All of these environments are characterized by rapid perturbations and typically by salinity changes, but also other ecological controls may generate stressful conditions that strongly affect the benthic biota. Ichnology has provided a powerful tool with which to identify these depositional settings by recognizing anomalous ichnofaunas (typical of marginal-marine brackish conditions), which display less variety and a lower abundance of forms than are found in fully marine environments (Buatois and Mangano, 2011).

Dalrymple et al. (1992) also classified estuaries into two main groups: wave-dominated and tide-dominated systems. In the former, there is a well-structured spatial distribution of energy. Three main zones are recognized: (1) the bay-head delta, a high-energy inner zone dominated by river processes; (2) the central basin, characterized by the mixing of marine energy and fluvial currents; and (3) the estuary mouth, dominated by marine processes.

Bay-head deltas are strongly stressful environments with unbioturbated or sparsely bioturbated deposits showing very low ichnodiversity, which is dominated by the dwelling structures of suspension feeders. In terms of ichnofacies, this zone mainly contains the *Skolithos* ichnofacies, followed by an impoverished *Cruziana* ichnofacies. Central basin settings show a combination of stress agents (brackish water, water turbidity, and oxygen depletion) associated with a low degree of bioturbation, although bioturbation may be moderate in some beds. The ichnofauna reflects the dominance of unspecialized deposit feeders and is characterized by the depauperate *Cruziana* ichnofacies, with minor contributions from the *Skolithos* ichnofacies. Although the estuary-mouth complex is highly variable, in terms of both trace concentrations and depositional settings, the bioturbation intensity and ichnodiversity generally range from moderate to intense (higher than in the previous zones), reflecting near-normal marine salinities; mixed depauperate *Cruziana* and *Skolithos* ichnofacies are present. In summary, trace fossil distributions along wave-dominated estuaries are mainly controlled by the salinity gradient, varying from the brackish waters of the inner zone to the near-open-marine salinity of the outer estuary.

Tide-dominated estuaries are characterized by a less pronounced distribution of energy along the estuarine valley because of the migration of intertidal runoff channels. Nevertheless, the following zones are recognized: (1) the upper estuary, a fluvio-estuarine transition zone characterized by freshwater conditions; (2) the middle estuary, meandering to straight tidal channels, tidal flats, and salt marshes; and (3) the lower estuary, comprising the outer zone with elongate subtidal sandbars, channels, and tidal flats (Figure 2).

Arthropods are the dominant tracemakers in the typical freshwater/terrestrial biotas of upper estuaries, and their activities are recorded in tidal rhythmites, which display a mixture of the elements of continental depauperate *Scoyenia* and *Mermia* ichnofacies. Farther towards the sea, the middle estuary commonly has brackish-water conditions. To different degrees in a number of settings, tidal flat deposits are dissected by a network of meandering tidal channels and creeks that migrate across the intertidal zone, producing lateral accretions in point bars (Dalrymple, 1992); the substrate-controlled *Glossifungites* ichnofacies may occur, corresponding to coplanar surfaces (incision during a sea-level fall and subsequent transgressive erosion), whereas mixed impoverished *Cruziana* and *Skolithos* ichnofacies record



Biogenic Sedimentary Structures, Figure 2 Reconstruction of a tide-dominated estuary from Santa Rosita Formation (Cambriano, Argentina) (From Buatois and Mangano, 2003, modified).

the activities of opportunistic communities that developed understressed conditions (brackish waters) in transgressive sediments overlying coplanar surfaces. The outer zone of the estuary displays fully or almost fully marine conditions, and the possible trace assemblages reflect the activities of organisms that include deposit feeders, predators, and suspension feeders in intertidal to subtidal settings. However, high-energy and rapidly migrating bedforms generally preclude the establishment of a mobile epifaunal and/or shallow infaunal biota (Buatois and Mangano, 2003).

Summary

Trace fossils can be retained as both paleontological and sedimentological entities because they represent not only the morphology and ethology of the tracemakers but also the physical characteristics of the substrate on which the tracemakers lived. In this sense, biogenic sedimentary structures can make meaningful contributions to numerous research fields in the earth sciences, with an integrated approach that articulates ichnological information with other sources of data. This is a good approach to reconstruct ancient depositional settings, which notably takes advantage of the integration of both sedimentological/stratigraphic and ichnological data. In marginal-marine environments (including estuaries), trace fossil assemblages play a major role in distinguishing open-marine, brackish-water, and freshwater/terrestrial deposits.

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Cross-references

Deltas
Estuarine Geomorphology
River-dominated Estuary
Sandflat
Soft Sediment Communities
Tidal Flat

BIOGENOUS SEDIMENT

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Synonyms

Biogenic sediments; Shelly sediments

Definition

Biogenous sediments are broadly defined as sediments consisting of large amounts of skeletal remains of macroscopic and microscopic organisms or remains of organic production.

Description

Estuarine sediments are derived from a number of sources including the watershed, continental shelf, atmosphere, erosion of the estuarine margins and bottom, and biotic activity within the estuary. The dominance of one sediment source depends on the interaction between the type and the quantities of available components. Terrigenous sediments such as sand and clay may be linked to riverine contributions, whereas biogenous sediments seem coupled to the lower estuary and the marine estuarine morphodynamic domains (Nichols and Biggs, 1985; Nichols et al., 1991). Biogenous sediments are formed from the insoluble remains of living organisms, such as shells, bones, and teeth (Davis, 1985; Cronin et al., 2003). They can be grouped in three major categories: calcareous biogenous sediments, siliceous biogenous sediments, and phosphatic biogenous sediments. The first group includes calcareous shells or remains of benthic organisms (mainly molluscs, snails, ostracodes, or foraminifera). The second group includes sponge spicules or diatoms and radiolarian remains, and finally, the last group includes fish scales and bones or organic matter formed in situ. These kinds of sediments are often used as a proxy of the human-induced changes in estuarine sedimentation (Colman and Bratton, 2003). Cronin (2007) shows that in estuarine environments such as the Chesapeake Bay diatoms can constitute 5–10 % of dry sediment, whereas calcareous shelly sediments can comprise as much a 5 %.

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Cross-references

Sediment Grain Size
Shell Beds

BIOINDICATORS

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Synonyms

Biomonitors; Ecological indicators; Environmental indicators

Definition

Bioindicators – biological attributes or characters of estuarine-associated organisms that are objectively or subjectively assessed to evaluate the conditions, status, or trends in the estuarine environment.

A broad range of biological attributes have been used as bioindicators in estuaries (Bortone, 2005). These biological attributes or characteristics can be selected from all levels of biological organization (with increasing order of specificity) from the community, population, and individual levels of biological organization at the individual level of organization; these finer aspects of biological organization include bioenergetics, reproductive, pathological, histological, physiological, immunological, genetic, biochemical, and molecular features. Generally, attributes at the higher levels of organization are more ecologically relevant but are of low specificity and sensitivity. Oppositely, attributes from lower levels of biological organization are less relevant ecologically but are of high specificity and sensitivity (Adams, 2002).

When selecting a biological indicator to assess estuaries, it is important to consider the time and space scale of response that would be useful for a particular situation (Bortone, 2008). For example, long-term (decadal), gradual changes in mean salinity within an estuary can be assessed using species distributions, their abundance, or community composition and diversity. Short-term changes in salinity might be better assessed using the physiological response (lethal dose or local movements) of individuals within a species. More specifically, changes to an individual's ability to osmoregulate (blood chemistry) would be a more immediate biological indicator of a situation of altered salinity. It is often preferable to measure several attributes in any given situation to serve as a corroboration and to allow assessment at several time and space units (Bortone et al., 2005).

Bioindicators can be either passive (e.g., observing growth) or active (extraction of tissues for chemical analysis). Not mutually exclusively, bioindicators can be sensitive to direct environmental stress (biochemical, physiological) or indirect stressors such as environmental changes that affect trophic and/or behavioral changes (Adams, 2005).

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BIOMAGNIFICATION

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Definition

Biomagnification is the process where chemical compounds are transferred from food to an organism resulting in higher concentrations compared with the source. It occurs when a chemical element or compound (chemical agent) then presents higher concentrations in the tissues of organisms as they occupy higher levels in the trophic web.

Fundamentals

Biomagnification is a phenomenon that occurs across different levels of the same trophic web and can involve whole populations and communities (Clark, 2001). It was first discovered when California brown pelicans were observed to have poor chick recruitment year to year due to the presence of $\sum$DDT in their tissues which is an endocrine disruptor interferring with Calcium fixation, and consequently egg shell's thickness and hardness. The $\sum$DDT was found to have originated in their main food resource (anchovies) which had fed plankton contaminated with $\sum$DDT from the Columbia River estuary that crossed pesticide-sprinkled agricultural areas. $\sum$DDT increased exponentially up to the female pelicans. Most organochlorines (e.g., PCBs) are now recognized as capable of undergoing biomagnification in aquatic environments. This phenomenon is closely related to polar food webs where large carnivores quickly acquire elevated concentrations of organic pollutants in their tissues that compromise their survival and progeny. Mercury and especially its organic forms also biomagnify in aquatic food webs (Clark, 2001; Costa et al., 2012). It is an ecological and analytical challenge to identify and quantitatively describe the biomagnification process across a given food web (Cardwell et al., 2013). Ideally, the trophic relations among the components of the food web should be well known, and analysis should be made in tissues from linked trophic positions. The study of the trophic transfer process along the food web is a useful tool to assess the biomagnification of trace elements from one trophic link to another (Cardwell et al., 2013). Also, biomagnification should preferably be confirmed by other analyses such as $\delta^{13}C$ and $\delta^{15}N$ isotopes.

One way to compare biomagnification across food webs is to plot the linear relationships between log chemical compound and $\delta^{15}N$ and use the regression slope (β) as a measure of the biomagnification power. The biomagnification power of a chemical compound is assessed using regression slope (β) of the simple linear regression, including all organisms of the food web possible: $\log[\text{chemical compound}] = \beta^*(\delta^{15}N) + a$, where a is the y-intercept. For mercury, the regression slope, i.e., biomagnification power, values range from 0.10 to 0.28 for tropical, temperate, and arctic marine and lacustrine ecosystems (Costa et al., 2012). This high range reflects the different composition of the food webs and/or differences in growth rate of organisms. On the other hand, the simple linear regression ($\log[\text{chemical compound}] = \beta^*(\delta^{15}N) + a$), including all organisms of the food web, is a useful tool to compare across habitats (pelagic, demersal, benthic) or ecological functions of the trophic web. It also assesses the bioavailability of a chemical compound to each organism. For example, the biomagnification power is higher for pelagic and benthopelagic species than for benthic species. It suggests that the chemical compound is readily available to the base of the benthic food chain but that trophic transfer is more efficient in pelagic and benthopelagic food chains (Costa et al., 2012). As a top consumer, human populations can often be involved in this environmental process when ingesting large predatory fish from both freshwater and marine origins. This constitutes a public health issue and must be seriously addressed by authorities (Costa et al., 2012).

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Cross-references

Bioaccumulation
Bioavailability

BIOMONITORS

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Synonyms

Sentinel organisms

Definition

Biomonitors are organisms that accumulate contaminants in their tissues and can be used to yield a relative measure of the total amount of contaminants in the environment integrated over a period of time. They respond simultaneously to different stressors, providing quantitative information on the quality of the environment.

Applications and characteristics

To observe the impact of anthropogenic activities on ecosystems and their development over a long period or different locations is a large-scale, costly, and time-consuming task. Monitoring such impacts is a challenge, once it involves systematic data acquisition in time and/or space in order to characterize distribution patterns and trends in all possible environmental compartments in which contaminants may accumulate (Chapman et al., 1982).

Biomonitors, by definition, are net accumulators of trace elements (Rainbow, 2002) and can be seen as self-contained, self-powered units that can respond to the presence of contaminants in the environment and are used for monitoring purposes around the world. Concentrations of contaminants in biomonitors are generally high enough to be easily measured with minor risk of contamination during sample collection or pretreatment when comparing to other environmental matrices, such as water samples. Moreover, the contaminants accumulated in biomonitors represent the most direct measure of bioavailable metal to an organism, i.e., the fraction of a contaminant that can be taken up from the environment and therefore with the potential to cause ecotoxicological effects (Rainbow, 2006; Luoma and Rainbow, 2008).

The first large-scale use of biomonitors was through the Mussel Watch Program, which developed monitoring activities using the blue mussel *Mytilus edulis* to quantify and assess spatial and temporal trends in coastal contamination of a suit of trace metals (Goldberg, 1986).

Several groups of organisms are currently used as biomonitors of environmental quality, including crustaceans, fish, corals, macroalgae, and benthic macroinvertebrates. Effectively used biomonitors facilitate comparisons of contaminants over different time and space scales. Across a pollution gradient, some organisms will be more tolerant and may become dominant, whereas the most sensitive groups may become rare. Important intra- and interspecific variation can be observed in the accumulation and tolerance of contaminants (organics or inorganics) in biomonitors, even for species belonging to the same taxonomic group (Amiard-Triquet et al., 2011). Therefore, it is advisable to use more than one biomonitor to increase the comprehension of different sources of contaminants (e.g., dissolved, particulate, sediments, etc.) (Luoma and Rainbow, 2008). It is also important to know the biology of each biomonitoring organism to understand the potential routes of metal uptake available to the organisms (Rainbow, 2006).

The most useful biomonitoring organisms are sedentary, abundant, and tolerant of environmental contamination and natural stressors. They should also be long lived to integrate variation in contaminant availability over a protracted period of time. They should also be large enough for analysis (Rainbow, 2006). Biomonitors must be resistant to handling during sample collection, manipulative experiments, and identification. Additionally, the more widespread the distribution of a biomonitoring organism, the greater its value as a cosmopolitan biomonitor providing cross-reference through large geographical areas (Rainbow and Phillips, 1993; Luoma and Rainbow, 2008).

Summary

Biomonitors are important tools to estimate and monitor the bioavailability of contaminants in the environment integrated over a specific period of time. The net accumulated contaminants may be used to identify ecologically significant pollutants.

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Cross-references

Bioavailability
Bioindicators
Biomagnification
Ecological Monitoring

BIOREMEDIATION

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Synonyms

Biotreatment of pollutants

Definition

Bioremediation refers to the use of an organism's metabolism to remove wastes, hazardous substances, or other pollutants. In general, microorganisms have been used as bioremediators, such as in phytoremediation, bioventing, bioleaching, landfarming, bioreactor, composting, rhizofiltration and biostimulation. However, not all contaminants are easily treated by bioremediation using microorganisms, and thus the elimination of a wide range of pollutants and wastes from the environment requires increased understanding of different pathways for specific bioremediation technologies and biotransformation processes.

Bioremediation options

Bioremediation has emerged as a promising technology, particularly as a secondary treatment option for oil cleanup. It has several potential advantages over conventional technologies, being less costly, less intrusive to the contaminated site, and more environmentally benign in terms of its end products (Zhu et al., 2004).

Bioremediation has been effectively used in estuarine environments as well as other aquatic ecosystems to remediate oils spills. It has proven to be an effective tool for also treating oil-contaminated marine shorelines. Microbes isolated from estuarine (brackish) waters have been of value in detoxification of many metals (Nagvenkar and Ramaiah, 2010).

In addition, bivalves have been utilized to mollify estuarine eutrophication by removing substances from the water column and reducing nitrogen (N) loads to coastal waters (Carmichael et al., 2012). Many molluscan species have the potential to reduce organic and inorganic compounds (nutrients) from aquaculture effluents; filter-feeding bivalves, microalgae, and macroalgae are potentially valuable organisms for reducing nutrient enrichment in estuarine and other coastal water bodies (Martinez-Cordova et al., 2011).

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Cross-references

Anthropogenic Impacts
Eutrophication
Macroalgae
Oil Pollution
Trace Metals in Estuaries

BIVALVE AQUACULTURE

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Definitions

Aquaculture is the farming of aquatic organisms by intervention in rearing to enhance production. It implies individual or corporate ownership of the stock.

Bivalve includes any member of the molluscan class Bivalvia, or Pelecypoda, characterized by having a two-piece (valved) shell.

Carrying capacity is the maximum population size or biomass that can be supported in a given area.

Epifauna are animals living on the surface of the sediments or hard substrate.

Infauna are animals living in the sediments such that the organism is entirely or nearly entirely covered.

Bivalve aquaculture classification

Bivalve aquaculture can be classified in two ways. The first focuses on the intended use of the final product, harvest, or restoration. Most bivalves are cultured for food, but some such as pearl oysters are cultured for jewelry, while others are produced to enhance or restore natural populations. Culture techniques for all uses are generally similar, but restoration stocks are maintained beyond normal harvest size to augment depleted populations. The major difference between harvest and restoration organisms involves the parental stock. Restoration stocks are generally selected to be genetically similar to the native populations to be restored, but, if disease is responsible for low population levels, it may be desirable to utilize stocks selected for disease resistance. Harvested individuals may be bred for genetic sterility, disease resistance, shape, meat yield, and fast growth.

An alternative classification scheme divides bivalves by habitat type: infaunal and epifaunal. Infauna includes those living near the surface (*Mercenaria*, *Cerastoderma*, *Meretrix*, *Ruditapes*), deeper burrowers (*Mya*, *Panope*), and the active burrowers (*Ensis*, *Solen*). Epifauna attach by cementing themselves to solid objects (oysters: *Ostrea*, *Crassostrea*, *Saccostrea*, etc.) or deploying a byssal thread (mussels: *Mytilu* and *Perna*). Others do not attach as adults, but move actively over the bottom (scallops: *Argopecten*, *Patinopecten*). The discussion below utilizes habitat classification because it facilitates discussion of environmental needs and the methods and equipment utilized during culture.

Hatcheries and nurseries

The two methods for obtaining seed for culturing are collection from wild stocks or the use of a hatchery. Culture historically started with species whose seed could easily be collected from the wild such as oysters, mussels, manila clams, soft-shell clams, and some scallop species. Wild-harvested seed is unavailable for some species such as the hard clam (*Mercenaria*) and the geoduck (*Panope*) because seed density is too low to support harvest. For species whose seed can be collected, the harvested seed are cultured in a manner similar to hatchery seed. When wild seed are unavailable, hatchery technology offers a means of obtaining seed. Hatcheries can also provide a more consistent seed supply and the opportunity for breeding and genetic improvement. Larger seed cost more but usually have higher survival. This cost dictates what sized juveniles must be produced to allow a reasonable trade-off between seed cost and survival of the planted crop.

Bivalve hatcheries are typically located on estuaries because waterfront access and reduced wave energy lower the cost of installing piping and pumps needed to provide water (there are exceptions, such as in Hawaii, where deep ocean water is available near shore, is high in nutrients and low in suspended sediments, and has constant temperature and salinity). Water pumped from estuaries has variable physical and chemical characteristics and often requires filtration and/or sterilization before use.

The hatchery process begins with ripening brood stock by warming the water and providing sufficient food, usually cultured microalgae (phytoplankton) although naturally available food can be utilized if water quality can be controlled. Most hatcheries begin the production season as the waters warm but may start earlier than nature so small seed are available to gain a growth advantage as the natural system warms. The early production of seed implies that the hatchery must maintain the newly set animals on cultured food for longer than is typical for seed produced later in the season. The conditioned brood stock is spawned, fertilized eggs collected, and the larvae are raised in tanks supplied with food, usually in the form of cultured microalgae. These larvae are held until they reach a size when they are ready to settle or set. Larval life span is determined by temperature, but for most species, the larval period is 10–20 days. Under optimal culture conditions, cultured species set at the lower end of the time spectrum. There are always exceptions to any generality dealing with bivalve culture. Instead of tanks and cultured algae, at least one commercial operation relies on lined open ponds and natural phytoplankton production for larval and nursery culture.

At setting, epifaunal and infaunal species may be treated differently. Epifauna such as oysters may be transferred to settling tanks and set as single animals or attached as clusters to shell or alternate materials. Scallops may be set on mesh or, as with infauna, allowed to attach by their byssus and then washed from the setting tank and placed in the appropriate containers. For many species, there is an intermediate nursery where late-stage larvae or immediate post-set are placed in a mesh-bottomed container and water, with cultured algae, is recirculated in the top and out of the bottom mesh (a downweller). Once the animals reach an appropriate size (typically about 0.75–1 mm), they are placed either on raceways or into upwellers (a mesh bottom cylinder where water flows up through the mesh, across the animals, and out through a pipe near the top). Scallops may be left in mesh bags and hung in a tank that is supplied with unicellular algae.

Once seed reach several mm in size, it is no longer economically feasible to culture algae for food, and the hatchery/nursery reverts to pumping water and food from the environment. Depending on location and species, these systems can be placed on land, or as floats in the water, but all are characterized by the use of some form of pumping mechanism to force water and food through the container of animals. If protected areas are available near a power source, floating systems can be utilized and pumping cost can be greatly reduced. If land-based systems are used, pumping costs are increased, but system security is improved. Epifauna may be kept in upwellers while infauna may be placed in raceways where sediments accumulate, helping to protect the seed from fouling. Fouling and overset control can also be achieved by coarsely filtering the water to remove potential fouling organisms, treating the tank and animals on a weekly basis with fresh water, air drying, or other methods. The treatment methods work well with species that tightly close (oysters), but cannot be utilized for species that cannot completely close their shells (scallops). As the animals grow, there is a constant need to increase both the space for the animals and the pumping capacity. These requirements mean that there is a trade-off between the maintenance and feeding requirements in the nursery and the potential losses incurred by planting small animals in nature.

To this point, the spatial area required for producing large numbers of animals is relatively modest. Most animals spawned in the spring will be placed in areas for grow-out in the fall. In some instances involving seed that did not reach planting size or when larger sized animals are needed, a nursery system that uses the passive movement of water through a cage (floating or on the bottom) or a nursery plot where animals are maintained at high density is utilized. The nursery plots are typically placed in easily accessed, sheltered locations. Nursery structures and grow-out structures (see below) are typically similar, but animals are usually at higher density and protection devices use finer mesh in the nursery.

Grow-out

Once animals reach a size where they can be planted for growth to market size, there are many methods depending on the needs of the organism, the environmental conditions, regulatory framework, and the value of the final product. Extensive methods (low density over a large area with minimal bed preparation and no predator protection), intensive methods (high-density bottom plantings with some form of predator protection), or "water column" methods (seed placed on long lines, hung from strings, placed in cages/trays or other containment vessels, or attached to stakes placed in the intertidal) are all utilized.

Epifaunal organisms may be placed in the inter- or subtidal directly on bottom beds that usually receive some preparation before planting. This may be rudimentary cleaning during the harvest of the prior crop or elaborate after harvest fallowing, followed by cleaning and resurfacing. In sites with large tidal amplitude, low earthen berms constructed on the tidal flats may be seeded or used to "finish" adults. These berms, topped by the incoming tide, retain water when the tide recedes, allow a longer feeding period, and ameliorate temperature fluctuations.

More intensive methods for epifauna utilize structures in the intertidal areas to contain the animals. These can be poles or stakes driven into the bottom and wrapped with mesh containing seed (bouchot culture for mussels) racks on which oysters are placed in bags arrayed horizontally (rack (or trestle) and bag culture) or seed may be set on stakes and that are placed horizontally (stick culture) in an intertidal area. Alternatively, cables can be stretched over the flats and baskets or cages attached to the lines (intertidal long-line culture). Cages offer both containment and some predator protection but must be maintained to prevent fouling from occluding the mesh. Structures in this zone must withstand storm and ice conditions, and an alternate site, in deeper water, a protected location, or a cool moist environment on shore, may be needed to provide protection. Fouling may be controlled by proper siting, turning the bags to expose the surface to the sun, power washing, other mechanical methods, or antifouling coatings. Rarely, chemical fouling control, such as dipping the containers and the animals in a brine solution, is utilized.

Beyond the intertidal zone, epifaunal species can be grown on line systems attached to rafts or a variety of surface floats (long-line culture) and may extend many meters below the surface. Mussels attach directly to the line systems, with intermediate supports to keep the crop from sliding off. Oysters and scallops may be attached to lines but are more typically placed in cages that are then attached to lines. Fouling control is an important maintenance procedure. Cages maintained on the surface can simply be inverted to allow the top to dry and the fouling organisms to die, while submerged cages must be cleaned or exchanged on a regular basis.

Infaunal species are typically planted in prepared areas (beds) in the intertidal or very shallow subtidal. Bed preparation may be rudimentary or elaborate. In most cases, protective mesh is stretched over the bed and its edges imbedded in the bottom to reduce predation. In areas of low predation, or when large seed are planted, the mesh may be eliminated. Mesh size is based on seed size. The mesh may be placed on the sediment surface and the seed allowed to dig through or the seed planted and the mesh placed over the seed. In the former, the seed must be smaller than the mesh, while in the latter, the seed are larger than the mesh. In both cases, the beds must be in areas of low wave energy or the mesh can be covered by moving sediments. Beds of shallow-burrowing species are typically mesh covered for the duration of the grow-out cycle (2–3 years) except in areas where ice can cause severe damage. In such areas, mesh may be removed from large seed (after the first summer's growth) in late fall and replaced in early spring. In ice-prone areas, meshes are maintained over small seed because of predation from ducks. Some high-value species such as the geoduck may be planted in tubes (PVC) implanted in the intertidal area and covered, individually or en masse, with mesh that may remain for several years. When the clams become larger and are deeply burrowed, the tubes and mesh are removed for final grow-out. In some areas, flats are bisected with low earthen berms being seeded to grow *Solen* without mesh. These berms retain the water for a longer portion of the tidal cycle.

Environmental effects

Environmental impacts of bivalve aquaculture have been shown to be relatively small and isolated because no food is added to the system. Further, bivalves filter the water, increase the biodeposition rate, and increase the rate of nutrient recycling, including denitrification. Exclusive of the potential for the importation of unwanted species, which has been reduced by importation regulations, the environmental impact of bivalve culture can be divided into three major categories: aesthetic, water column, and benthic. Aesthetic effects have caused delays in obtaining permits for farms because property owners do not want to see culture gear or hear noise associated with gear maintenance and harvest. Proper siting and education of nearby property owners and culturist usually result in compromises that satisfy both parties. Water column effects are generally positive because water clarity is improved by removal of inert particles and microalgae. Too many bivalves placed in the water column can reduce growth rates because the local carrying capacity is exceeded. In temperate areas, annual periods of low temperature plus low growth may add substantially to the length of the culture cycle. Studies documenting where ecosystem carrying capacity has been exceeded have recently been reviewed (Burkholder and Shumway, 2011). The culture of infauna and the bottom culture of epifauna typically results in fewer water column effects than the epifaunal culture on long lines or rafts because bottom culture is conducted in a single layer and results in less biomass per square meter than water column methods.

The biggest environmental change caused by bivalve aquaculture is benthic due to the accumulation of biodeposits on the sea floor that in turn can affect the other benthos. For animals cultured in the water column, biodeposits can greatly exceed normal deposition by animals living in or on the bottom. This accumulation and its effects were documented over a half century ago (Ito and Imai, 1955; see also Norkko and Shumway, 2011), and effects can be reduced by proper siting or site rotation. For infaunal and epifaunal benthic culture, the biodeposits are limited by food supply and resuspension/erosion rates. If the food supply is too low, growth decreases and deposition of feces and pseudofeces decreases. If food supplies are not limiting, siting the culture in an area of moderate currents can reduce excessive buildup of biodeposits. This scouring effect is particularly evident in intertidal or shallow subtidal culture areas where both currents and waves serve to clear the bottom. In spite of this natural sediment movement, the increased density of cultured organisms causes an increase in the fine particle content of the sediments. This change plus the physical presence of the cultured species can alter the infaunal community. Protective structures such as mesh increase epibiota and may emulate the structure and function of nearby reef or seagrass areas. In general, except for the increased density of the cultured species and effects associated with harvesting, bottom culture of bivalves has relatively little ecosystem level effect (Dumbauld et al., 2009). Studies on effect of the adding structural components (PVC tubes) for geoduck culture on the US west coast and screening for clam culture on the US east coast have found that these culture operations do not significantly alter the ecosystem processes (Kraeuter et al., 2013; Van Blaricom et al., 2013).

Breeding

Selective breeding has been conducted on a few bivalve species. Oysters have received the most attention because of the need to develop strains that resist diseases. Stocks of *Crassostrea virginica* have been developed that are resistant to MSX (*Haplosporidium nelson*)

(Haskin and Ford, 1979). These stocks have been further bred to provide lines that are better suited for certain regional conditions. In addition, oysters (*Crassostrea gigas* and *C. virginica*) have been subject to ploidy manipulation to provide for animals with reduced or no gonadal development allowing marketing a uniform product on a year-round basis. By developing tetraploid (four sets of chromosomes) stocks (Allen and Guo, 1998), hatcheries are now able to provide triploid (functionally sterile) oysters for the culture market. Some breeding work has been conducted with hard clams (*Mercenaria mercenaria*), but most of this was to develop faster, more uniform growth. There is evidence that some strains of hard clams are better suited for certain environments and have higher resistance to the disease QPX (Quahog Parasite Unknown) (Ragone-Calvo et al., 2007; Kraeuter et al., 2011), but the stocks have not been bred for these characteristics.

Health effects

An important aspect of bivalve culture is the requirement for high water quality. As bivalves filter the water, they concentrate microorganisms. This characteristic, and because many bivalves are eaten without cooking, means they must be cultivated in waters free of organisms that cause human sicknesses. This constrains site selection and means the presence of bivalve culture provides an incentive for water quality managers to maintain or improve bacterial water quality.

Summary

Over 75 % of the bivalves harvested from estuaries are produced by aquaculture which is rapidly increasing (Creswell and McNevin, 2008; Rheault, 2012). Bivalve aquaculture does not rely on adding feed to the environment, and as such is considered to be a form of nutrient extraction. If populations are dense enough, they can become a natural control of eutrophication (Cloern, 1982; Officer et al., 1982). In addition, since bivalves are filter feeders, they remove fine particles from the water and can increase water clarity. Through biodeposition, they enhance nutrient recycling including denitrification (Newell et al., 2005). Lastly, sites culturing bivalves for human consumption require the highest water quality standards and can provide important incentives for increasing or maintaining estuarine water quality.

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Cross-references

Shellfish Production

BIVALVE MOLLUSCS

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Synonyms

Molluscs; Pelecypods

Definition

Bivalves (nearly 20,000 species) are one class in the phylum *Mollusca* (Abbott, 1974; Gosling, 2003; Gofas, 2013). They secrete a relatively hard shell that covers the mantle and gill tissues. The shell grows out from the point of articulation, the hinge, with new layers regularly added from the mantle tissues. Some species live free, singly (clams), or in dense aggregations (scallops); others live attached to each other by either byssal threads (Wilker, 2011) or cement (Burkett et al., 2010; Moeller and Matyjaszewski, 2012). Many serve as economically important wild stock fisheries or aquaculture species (see Gosling, 1992, 2003; Spencer, 2002; Hardy, 2006; Shumway and Parsons, 2006; FAO, 2009, see also FAO and NMFS websites). Most are filter feeders (Dame, 1993, 1996; Wildish and Kristmanson, 1997) or deposit feeders (Rhoads, 1973; Kamermans, 1994), but some species are very specialized (Abbott, 1974), boring into wood, rocks, corals, and even other bivalve species (Families *Teredinidae*, *Pholadidae*, some *Mytilidae*, and *Veneridae*). Clams and mussels (*Mytilus*, Gosling, 1992), for example, are found from full strength salinities, in estuaries to freshwater (perhaps a third of all bivalve species, see Haag, 2012) (Dame, 1996; Levinton, 2013). Many freshwater clams and mussels reside in isolated water bodies, have atypical larval adaptations, and are currently endangered (Haag, 2012).

Characteristics

Estuaries and their component habitats are generally recognized as some of the most productive and important ecosystems, as they provide critical feeding, spawning, and nursery areas for numerous species, including economically important fish, shellfish, and waterfowl, in addition to ecologically valuable invertebrate and vertebrate species (Boesch and Turner, 1984; Beck et al., 2001, 2003; Barbier et al., 2011). They are also one of the most impacted ecosystems on the planet (Lotze et al., 2006; Airoldi and Beck, 2007; Molnar et al., 2008; Beck et al., 2009, 2011). One important and common species lineage is the bivalve molluscs, found both intertidally and subtidally in estuaries. Habitat-forming bivalve species (ASMFC, 2007) might be viewed as those that are (1) “reef-forming” (see DeAlteris, 1988; Waldbusser et al., 2013, Figure 1), (2) “aggregation-forming,” or (3) “shell-accumulating.” Many species are or were important economically, including clams, scallops, mussels, and oysters (MacKenzie, 1996, 1997a, 1997b, 1997c; Bell et al., 2005). For example, the softshell clam (*Mya arenaria*) once supported a commercial fishery in the Chesapeake Bay that is currently nonexistent (Abraham and Dillon 1986). Restocking or restoring these invertebrates can be very different from conventional finfish approaches (Breitburg et al., 2000; French McCay et al., 2003; Bell et al., 2005; Arnold, 2008; Beck et al., 2009; Coen et al., 2011a).

Many “free-living” (e.g., non-reef-forming or solitary) species occur in coarse sand to “shelly” habitats cobbles or in submerged aquatic vegetation (SAV) which can provide some protection from predators (Sponaugle and Lawton, 1990; Irlandi, 1994; Kraeuter and Castagna, 2001; Grabowski, 2004; Grabowski et al., 2008). Many clam species (e.g., hard clams, *Mercenaria* spp., softshell clams *Mya*, etc.) that occur infaunally are able to “migrate” horizontally, as well as vertically, when conditions are adverse (Newell and Hidu, 1986; Dame, 1996) such as low dissolved oxygen (Diaz and Rosenberg, 1995; Burnett, 1997), whereas cemented species are unable to relocate (Lenihan and Peterson, 1998; Lenihan et al., 1999; Altieri and Witman, 2006; Breitburg et al., 2009; Johnson et al., 2009). Loss of dense populations of bivalves can have significant ecosystem effects (Altieri and Witman, 2006; Beck et al. 2011).

Harvesting (=disturbance) of bivalves and associated faunas in sediment or reefs (Hall et al., 1990; Coen, 1995; Dayton et al., 1995; Thrush et al., 1995; Lenihan and Micheli, 2000; Stokesbury et al., 2011) can have significant effects on their functioning and recovery (Hall, 1994; Coen and Luckenbach, 2000; Jackson et al., 2001; French McCay et al., 2003; Lotze et al., 2006; Grabowski and Peterson, 2007; Grabowski et al. 2012).

Mobile infaunal species are found in a variety of substrates including sand, mud, shell, and mixtures of these (Dame, 1996; Levinton, 2013). For example, razor clams (family *Pharidae*) can move very rapidly in estuarine sediments with a specialized shell and foot. Non-cementing scallop species are relatively mobile as juveniles and adults moving off the bottom for short excursions (Fay et al., 1983; Shumway and Parson, 2006), for example, to flee predators (Pohle et al., 1991; Ambrose and Irlandi, 1992; Arnold, 2009). Some species have siphons that are used for feeding and respiration. These straw-like structures also allow many species to reside deeper in the sediment (soft clams, *Mya arenaria*; see Figure 2) providing some protection from both lethal and sublethal predators (Irlandi, 1994). However, these fleshy tissues are then available to “sublethal” predators whose diets can be

Bivalve Molluscs, Figure 1 Examples of intertidal and subtidal shellfish habitats. (**a** and **b**) Pen shell, *Atrina zelandica*, aggregations in New Zealand (Source: Simon Thrush, University of Auckland, New Zealand); (**c**) *Modiolus modiolus* assemblages in St. Joe Bay, Florida, USA (Source: Brad Peterson, State University of New York, Stony Brook); (**d**) nesting oyster catchers on intertidal shell accumulations (racks) along the Intracoastal Waterway, SC, USA (Source: Phil Wilkinson, South Carolina Department of Natural Resources); (**e**) *Geukensia demissa and Crassostrea virginica* among *Spartina* stems in New Jersey, USA (Source: David Bushek, Rutgers University); (**f**) dense pen shell aggregation in a seagrass bed in the intertidal zone in Dubai (Source: Raymond Grizzle).

Bivalve Molluscs, Figure 2 *Mya arenaria* (soft clams) with extended fleshy siphons extended. The current softshell fishery in the Chesapeake Bay (USA) is nearly extirpated. This is another example of a bivalve species that once supported a commercial fishery that is no more. http://dnr2.maryland.gov/fisheries/Pages/shellfish-monitoring/clams.aspx.

Bivalve Molluscs, Figure 3 *Rangia cuneata* clam shells Texas (USA) coast (Photo by Steve Black. See http://www.texasbeyondhistory.net/coast/prehistory/images/shellfish.html).

dominated by cropped tissues (Peterson and Quammen, 1982; Lindsay et al., 1996; Meyer and Byers, 2005).

Another common estuarine to marine bivalve, pen shells (family *Pinnidae*) are relatively large bivalves that bury themselves partly into the substrate and are anchored by byssal threads. Only the upper portion of the shell is exposed above the sediment (referred to as "emergent shellfish beds"; see ASMFC, 2007), providing additional habitat (Figure 3a, b) for other organisms, when either live or dead (Keough, 1984; Kuhlmann, 1998; Cummings et al. 1998, 2001; Munguia, 2004). In dense numbers, these live and dead pen shells create a critical habitat in many systems (Connell and Keough, 1985; Munguia, 2004). In some areas stranding events point to large nearshore populations such as those near Sanibel Island, Florida, USA (L. Coen personal observations. Perry, 1936). Many bivalve "foundation species" support enhanced diversity quite often and complex communities (Altieri and Witman, 2006).

Many smaller clams such as the estuarine wedge clam, *Rangia cuneata* (Figure 3), form dense filtering aggregations in brackish to estuarine salinities with regular freshwater input. These bivalves, as well as many others, serve as important food sources for fish, crabs, and birds (LaSalle and de la Cruz, 1985; Ruiz, 1987; Ruiz et al., 1989; ASMFC, 2007). Mined deposits of wedge clam shells from Lake Pontchartrain, Louisiana, from 1933 to 1990 (Abadie and Poirrier, 2000) supported the wild stock oyster industry in Louisiana. The shells were planted in estuaries on state and leased grounds until a moratorium stopped the removal of the natural clam shells for their intrinsic functions (ASMFC, 2007). Mined oyster shell has been dredged also from many estuaries throughout the USA for use in replanting leased or state-managed shellfish "grounds" (Hargis and Haven, 1999; Burrell, 2003).

Shell mounds or "middens" from indigenous peoples are found in nearly all coastal areas where bivalves were once common or still are (Ceci, 1984; Beck et al., 2009; Balbo et al., 2011). Along the Gulf of Mexico and Atlantic coasts of the USA, middens primarily consist of *C. virginica*, but also clams (*Mercenaria* spp., *Rangia*), blue (*Mytilus* spp.) and ribbed mussels (*Geukensia* spp.), and slipper snails (*Crepidula* spp.) (Mackenzie et al. 1997a; Saunders and Russo, 2011). For North America, European settlers (Dutch, English, and French) began to harvest these species in the 1600s. In colonial days, bivalves were quite abundant (Kent, 1992; MacKenzie, 1996; Mackenzie et al., 1997a; Kirby, 2004), but through the late nineteenth century on, stocks in North America and many other areas became depleted (Rothschild et al., 1994; Kirby, 2004; Beck et al., 2009; zu Ermgassen et al., 2012).

Even when dead, bivalve shells accumulate (intact or broken as "shell hash," "rakes") in or on the sediment floor often in sufficient quantities to provide significant structure and habitat for a variety of organisms (Anderson et al., 1979; Lehnert and Allen, 2002; Street et al., 2005; Coen et al., 2006, 2011a; ASMFC, 2007; Summerhayes et al., 2009). In some areas, boat wakes have apparently degraded the natural reefs resulting in large accumulations of dead shell along the shorelines (Grizzle et al., 2002; Wall et al., 2005). Oystercatchers and other wading birds use intertidal to supratidal shell accumulations as nesting/feeding sites along dredged areas such as the Intracoastal Waterway (ICW) (Figures 1d and 4a, b, Marsh and Wilkinson, 1991; Goss-Custard, 1996; ASMFC, 2007; Sanders et al., 2008; Thibault et al., 2010). This can even occur when nonnative bivalves (*Mya*) are introduced into novel estuarine habitats (e.g., Dumbauld et al., 1993). Shell of many different bivalve species [mined from seafloor or from middens (see above)

Bivalve Molluscs, Figure 4 (**a**) Washed intertidal shell (racks) in South Carolina, USA along the IWW (see also Anderson et al. 1979, Source: Felicia Sanders, SCDNR, Charleston, South Carolina, USA). (**b**) Oystercatchers feeding in Cape Romain, SC, USA (Source: Felicia Sanders, SCDNR, Charleston, South Carolina, USA).

or accumulated at shucking houses] is used for rehabilitation and restoration of other bivalves (LaSalle and de la Cruz, 1985; Kraeuter et al., 2003; Waldbusser and Salisbury, 2014).

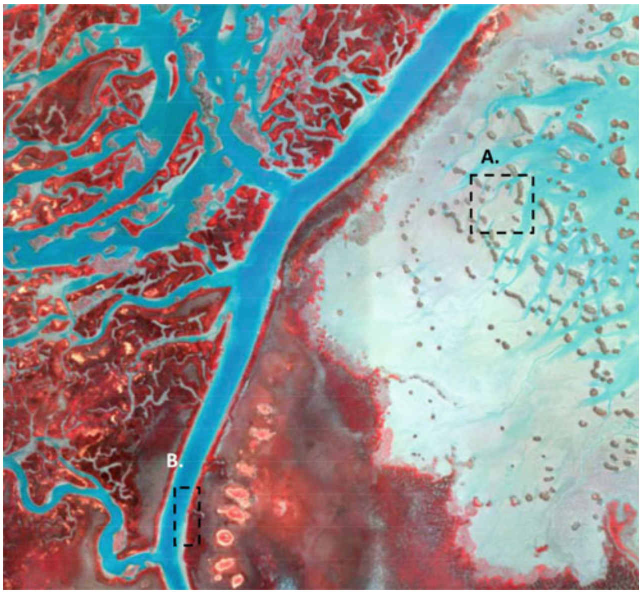
Many bivalve species, especially the reef-forming oysters (e.g., the True Oysters, *Ostreidae*, genus *Crassostrea*; see Carriker and Gaffney, 1996), are under pressure or have already been impacted significantly across the globe (e.g., Rothschild et al., 1994; Lotze et al., 2006; ASMFC, 2007; Beck et al., 2009, 2011, zu Ermgassen et al., 2012). It is these reef-forming species (often called "ecosystem engineers," Gutiérrez et al., 2003; Byers et al., 2006) that have been the focus of recent and current restoration efforts (Beck et al., 2011; Powers and Boyer, 2014), especially for their "ecosystem services" in North America (e.g., Coen et al., 1999a, 2007; Coen and Luckenbach, 2000; ASMFC, 2007; Grabowski and Peterson, 2007; Grabowski et al., 2012; Brown et al., 2014; La Peyre et al., 2014b).

One widely ranging species, the Eastern oyster, *Crassostrea virginica*, forms living subtidal and intertidal biogenic reefs that are a dominant feature of many Atlantic and Gulf US coastal estuaries (Chestnut, 1974; DeAlteris, 1988; ASMFC, 2007; Beck et al., 2011). Because of its extensive range and importance as a major fishery species in the USA dating back to the late eighteenth century (Brooks, 1891), there exists an extensive body of information on the biology of this species and their populations (Marshall, 1954; Galtsoff, 1964; Bahr and Lanier, 1981; Sellers and Stanley, 1984; Stanley and Sellers, 1986; Kennedy et al., 1996). However, its populations have declined significantly in many US estuaries that once had major fisheries (Rothschild et al., 1994; Kirby, 2004; NRC, 2004; Lotze et al., 2006). The causes are numerous and interrelated including overharvesting, pollution and related impacts, habitat destruction, and oyster diseases. Most harvestable oyster populations were primarily subtidal (Figures 5 and 6), such as those in the Chesapeake Bay (Maryland and Virginia), Delaware Bay (Delaware and New Jersey), and the Gulf of Mexico (Florida to Texas) (MacKenzie, 1996; MacKenzie et al., 1997a; zu Ermgassen et al., 2012).

In contrast, many intertidal *C. virginica* reefs (Figures 7 and 8) such as those in the southeastern (Galtsoff, 1964; Bahr and Lanier, 1981; ASMFC, 2007) and southwestern USA develop in locations where salinities is often moderately high, water column and resuspended food are sufficient, and siltation is not excessive, although most oysters can thrive in highly turbid waters (Coen, 1995). In these areas intertidal oysters often grow in isolated patches

Bivalve Molluscs, Figure 5 Image of a shallow subtidal *Crassostrea virginica* restored reef from Chesapeake Bay, MD, USA (Source: K. Paynter, University of Maryland, College Park).

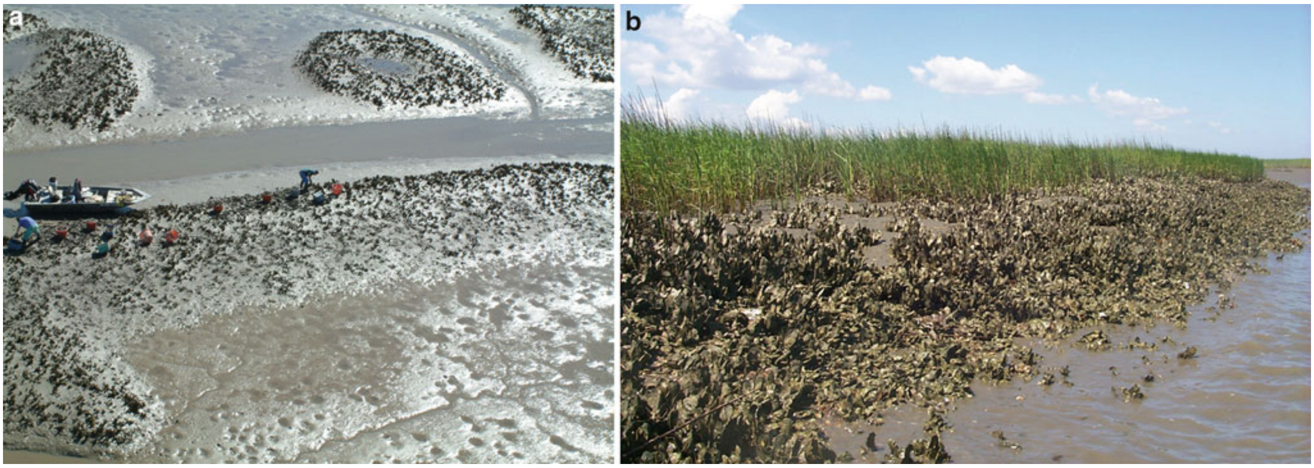
Bivalve Molluscs, Figure 6 Restored oyster reef in the Great Wicomico River on the western shore of lower Chesapeake Bay, USA. The high-relief reef harbored about 1,000 oysters m^{-2} of four age classes (Schulte et al. 2009) and is thought to resemble historical reefs from Colonial times (Source: R.P. Burke and R.N. Lipcius, VIMS, VA, USA, image taken from Remotely operated underwater vehicle or ROV video).



Bivalve Molluscs, Figure 7 Multispectral aerial image of intertidal oyster reef types typical of southeastern USA. The dashed square on the right (see Inset A, Figure 8), "oyster flats" in embayments, and the dashed rectangle (see inset B, Figure 8) represents "fringing oyster reefs" adjacent to salt marsh-lined tidal creeks (see ASMFC 2007 and SCDNR, 2008, for more information; Charleston Co., South Carolina, USA).

away from shore-lines (Figures 7a box and 8 inset a) or along fringing marsh (*Spartina*), on mangroves and around their islands, bordering creeks, rivers, sounds, and embayments (Figures 7b rectangle and Figure 8 inset b, Galtsoff, 1964; Bahr and Lanier, 1981; Burrell, 1986; ASMFC, 2007; Volety, 2013; Baggett et al., 2014). Mussels (e.g., *Geukensia* spp.) also can be quite abundant (Figures 1c, e, and Figure 9) in fringing marshes and intertidal and subtidal natural and restored oyster reefs (Bertness and Grosholz, 1985; Franz, 2001; Luckenbach et al., 2005; Walters and Coen, 2006).

Grabowski and Peterson (2007) and others (Coen et al., 1999a; Coen et al., 2007; Baggett et al., 2014) have delineated ecosystem services provided by oyster reef habitats: (1) oyster production; (2) water filtration/fecal concentration; (3) nutrient sequestration; (4) habitat for fish and invertebrates and augmented production; (5) stabilization of adjacent habitats/shorelines; and (6) enhancement of ecosystem complexity. Recent research has attempted to quantify the contribution of oyster habitats to ecosystem functioning in economic terms (Peterson et al., 2003; Grabowski and Peterson, 2007; Grabowski et al., 2012). For example, oysters create vertical, three-dimensional reef or bed habitats utilized by numerous fishes, crustaceans, other invertebrates, birds, and mammals (reviewed in Coen et al., 1999a; Coen et al., 2007; ASMFC, 2007). The abundances and biomasses can rival SAV, salt marshes, or mangroves in terms of harboring



Bivalve Molluscs, Figure 8 Detail of square, (see Figure 7a) of a typical "oyster flat" area (Source: SCDNR) in southeastern USA. Detail of rectangle, (see Figure 7b) of a typical "fringing oyster" marsh lined tidal creek (Source: Loren Coen) in southeastern USA.

Bivalve Molluscs, Figure 9 Mussels (*Geukensia demissa*) can be quite abundant in fringing marshes and intertidal and subtidal natural and restored oyster reefs (Source: David Bushek, Rutgers University, NJ, USA).

organisms (Glancy et al., 2003; Tolley and Volety, 2005; Coen et al., 1999a; Coen et al., 2006; Hosack et al., 2006; Rodney and Paynter, 2006; ASMFC, 2007; Coen et al., 2007; La Peyre et al., 2014b). Both subtidal (Figures 5 and 6) and intertidal (Figures 7 and 8) oyster habitats can support a diverse suite of sessile and mobile species (over 300 species in North Carolina; Wells, 1961). Natural reefs support greater numbers than the surrounding natural sand, mud, or even marsh habitats (Coen et al., 1999a; Glancy et al., 2003; Plunket and La Peyre, 2005; Coen et al., 2006; Hosack et al., 2006; ASMFC, 2007; Shervette and Gelwick, 2008; Taylor and Bushek, 2008; Stunz et al., 2010; Humphries et al., 2011a; Humphries et al., 2011b; Shervette et al., 2011). Constructed subtidal and intertidal reefs can also support diverse communities throughout *C. virginica*'s range (e.g., 115 macrofaunal species in South Carolina, Coen et al., 2006; see also Harding and Mann, 1999; Rozas and Zimmerman, 2000; Luckenbach et al., 2005; Tolley and Volety, 2005; Rodney and Paynter, 2006; ASMFC, 2007; Taylor and Bushek, 2008; Gregalis et al., 2009; Stunz et al., 2010; Kingsley-Smith et al., 2012; Brown et al., 2014).

Numerous studies have documented positive synergies between bivalves (especially mussels and oysters) and other habitats such as seagrass (Figure 1c) (Valentine and Heck, 1993; Everett et al., 1995; Peterson and Heck 1999; Peterson and Heck 2001a; Peterson and Heck 2001b; Wall et al., 2008, 2011; Booth and Heck, 2009). This largely results from improved water clarity from bivalve feeding activities thereby increasing light. Water flows are also slowed and sediments and seeds fall out around the reefs. Shellfish release ammonia also and other metabolites and nutrients for SAV (Williams and Heck, 2001). Native oysters and bivalve aquaculture may potentially play a parallel role with SAV (Newell, 2004; Erbland and Ozbay, 2008; Dumbauld et al., 2009; NRC, 2010, Coen et al. 2011a), enhancing or protecting other habitats from erosion (Meyer et al, 1997; Coen et al., 2004, 2007; Piazza et al., 2005; Beck et al., 2009). One of the direct and indirect influences of shallow subtidal or intertidal oyster shell (reef) construction is protection or enhancement of fringing marsh habitats (e.g., Meyer et al, 1997; Piazza et al., 2005; Currin et al., 2010; Scyphers et al., 2011). "Living shorelines" are one set of approaches (Figure 10) that may provide an alternative to stabilization with hardened structures (bulkheads, revetments, concrete) which have armored major portions of estuarine shorelines (Douglass and Pickel, 1999; Scyphers, 2012). Their use attempts to minimize the relatively poor habitat quality along developed shorelines (e.g., Seitz et al., 2006). Landscape issues

Bivalve Molluscs, Figure 10 Development of ReefBLK living shoreline reefs constructed of rebar and filled with shell in Texas, USA. Natural recruitment and growth after roughly one year (Source: Jeff DeQuattro, TNC, Mobile, AL, USA). See http://www.reefblk.com/ for more information.

Bivalve Molluscs, Figure 11 Intertidal *Ostrea lurida* beds in Port Eliza, Nootka Sound, Vancouver Island, Canada (Source: B. Kingslett, Deep Bay Field Station, Vancouver Island University, BC, Canada).

are critical to consider since mobile fauna (fishes and invertebrates) use multiple habitats either for feeding, refuge (Micheli and Peterson, 1999; Harwell et al., 2011), or because they must move with tidal exposure (Coen et al., 1999a, 2006; ASMFC, 2007).

Shell alone once planted or aquaculture gear once deployed (Erbland and Ozbay, 2008; Dumbauld et al., 2009; Marenghi and Ozbay, 2010; Coen et al., 2011a) immediately attracts a diverse assemblage of organisms prior to oysters and other sessile organisms recruiting (Luckenbach et al., 2005; Walters and Coen, 2006). Mobile resident and transient species can be found immediately on these "artificial reef" structures (Dumbauld et al., 1993; Wenner et al., 1996; Coen et al., 1999b, 2006; Lehnert and Allen, 2002; Tolley and Volety, 2005; ASMFC, 2007; Gregalis et al., 2009; Humphries et al., 2011a; Humphries et al., 2011b). With time, oysters and mussels and other filter-feeding invertebrates (barnacles, cnidarians, tunicates) (Haven and Morales-Alamo, 1966; Dame et al., 2001; Newell, 2004; Luckenbach et al., 2005; Walters and Coen, 2006; Coen et al., 2007; Kellogg et al., 2013) then settle. Cumulatively, these filter feeders can filter significant quantities of water, potentially improving water clarity/quality locally (Nelson et al., 2004; Newell, 2004; Grizzle et al., 2006; Grizzle et al., 2008a; zu Ermgassen, 2013a, b; La Peyre et al., 2014b) through increased denitrification rates and enhanced nutrient sequestration into the shells themselves (Piehler and Smyth, 2011; Carmichael et al., 2012; Higgins et al., 2011, 2013; Kellogg et al., 2013; Smyth et al., 2013; Hollein and Zarnoch, 2014). They also form a unique association with fringing salt marsh habitats where the two habitats often abut (Meyer et al., 1997; DeBlieu et al., 2005; Piazza et al., 2005; Coen et al., 2006, 2007, 2011b). The other ecosystem services discussed above are just coming into play outside of North America, based on publications and presentations at meetings such as International Conference on Shellfish Restoration (ICSR, Coen pers. obs., http://www.oyster-restoration.org/workshops-meetings-related-to-oyster-restoration/).

Similarly on the west coast of the USA, the native oyster, *Ostrea* spp. (Figures 11 and 12; Polson and Zacherl, 2009; Polson et al., 2009) populations have reached near extirpation (Trimble et al., 2009; Beck et al., 2009, 2011, zu Ermgassen et al., 2012), with perhaps a few examples in Canada of what their beds once resembled (Jacobsen, 2009). This species never probably formed high vertical relief reefs, but rather abundant "beds" both intertidally (see Figures 11, 12) and subtidally (Beck et al., 2009; Polson and Zacherl, 2009; Trimble et al., 2009; Baggett et al., 2014).

Nonnative bivalve species (e.g., *Crassostrea* spp.) introductions, either through direct and accidental introductions, were first penned by Elton (1958) and are having significant and complex impacts on worldwide (Figures 13 and 14) (Wolff and Reise, 2002; Ruesink et al., 2005;

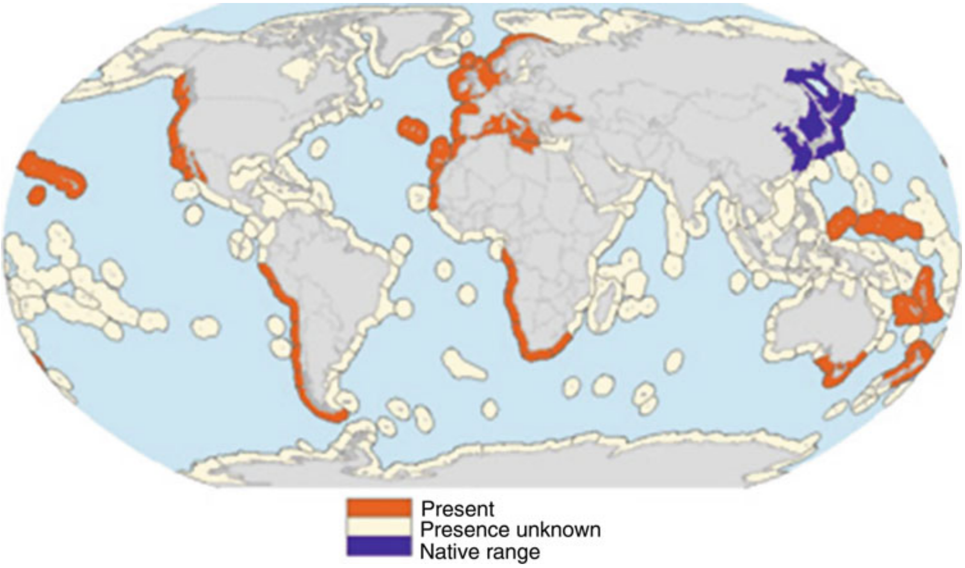
Thieltges et al., 2006; Decottignies et al., 2007; Thomsen et al., 2007; Brandt et al., 2008; Molnar et al., 2008; Kochmann et al., 2008; Wrange et al., 2010; Padilla et al., 2011). In Europe, the native flat oyster, *Ostrea edulis* has been replaced by the introduced Japanese oyster, *Crassostrea gigas* (see Figures 13, 14a, b). The same situation has occurred in many other estuaries throughout the world (Ruesink et al., 2005; Molnar et al., 2008; Kimbro et al., 2009) where native species have declined (Beck et al., 2009) and nonnatives have been introduced to support a commercial fishery (NRC, 2004).

Bivalve Molluscs, Figure 12 *Ostrea lurida* from above beds (see Figure 11) in Vancouver Island, Canada (Source: B. Kingslett, Deep Bay Field Station, Vancouver Island University, BC, Canada).

In many nearshore and estuarine areas, introduced oysters are transforming the landscape in manifest ways (reviewed in Ruesink et al., 2005; Smaal et al., 2005; Molnar et al., 2008; Padilla et al., 2011). In the Wadden Sea, for example (Nehls and Büttger, 2007), the invasion of the Pacific oyster *Crassostrea gigas* is causing major habitat shifts from the formerly dominant native bivalve, the blue mussel, *Mytilus edulis* which formed beds to intertidal oyster reefs (Figures 14a, b). The consequences for native benthic communities, mussel-eating birds and other higher food web consumers, the mussel fisheries, etc. have yet to be resolved. In some cases diseases (MSX) or hitchhikers (slipper shells, *Crepidula* spp.) have had significant impacts (Elton, 1958; Burreson and Ford, 2004; Decottignies et al., 2007).

Restoration

As mentioned already, past oyster restoration efforts have focused on recovering lost or impaired oyster fisheries (MacKenzie, 1996; MacKenzie, 1997a, MacKenzie, 1997b, MacKenzie, 1997c, Beck et al., 2009, 2011). Because of the significant decline of oyster reefs worldwide (e.g., Beck et al., 2009, 2011) and related efforts to reconstitute these species' habitats, numerous attempts (from small to large scale) have been initiated, especially in North America for non-resource-related ecosystem functions (Coen and Luckenbach, 2000; Gutiérrez et al., 2003; Coen et al., 2007; Brumbaugh and Coen, 2009; La Peyre et al. 2014; Powers and Boyer, 2014). As mentioned above, the focus of recent (since 1990s) enhancement and restoration efforts has been for the other "ecosystem services" (Luckenbach et al., 1999; Coen and Luckenbach, 2000; Grabowski and Peterson, 2007; Baggett et al., 2014;



Bivalve Molluscs, Figure 13 Relatively recent distribution of the Pacific oyster, *Crassostrea gigas*. Orange is its "invasive" (nonnative) range, blue is its "native" range (prior to reanalysis by molecular approaches), and white is unknown (potential) occurrence from Molnar et al. (2008, see Figure 4).

Bivalve Molluscs, Figure 14 (**a**, **b**) Pacific oyster, *Crassostrea gigas* replacing mudflats and cockle/mussel areas in the Dutch Wadden Sea. Above – (**a**), mudflats and (**b**) typically what invaded flats look like after five years. Both images taken in 2013 (Source: Carola van Zweeden, Institute for Marine Resources and Ecosystem Studies (IMARES), Centre for Shellfish Research, Netherlands).

Powers and Boyer, 2014) versus stock enhancement (Bell et al., 2005; Luckenbach et al., 2005; Arnold, 2008). As described elsewhere, we define restoration as "The process of establishing or reestablishing a habitat that in time will come to closely resemble a natural condition in terms of structure and function" (see Coen and Luckenbach, 2000; Coen et al., 2004; Baggett et al., 2014). One of the key differences among sites has been either a deficiency of adequate (=appropriate) shell or other hard substrate for settlement (Figure 15a, b) or a limitation of oyster larval recruits ("spat"). Modeling is beginning to get at those sites that have enhanced larval recruitment (e.g., Kim et al., 2013). In areas where larval supply is limited (e.g., Hudson River Estuary; Levinton and Waldman, 2011; Starke et al., 2011; Levinton et al., 2012; Grizzle et al., 2013), spat on shell ("SOS") is one approach. Shell ("cultch") with small set oysters ("spat") either from hatcheries or from field sets can be required (Figures 16a, c) (Coen and Luckenbach, 2000; Coen et al., 2004; Brumbaugh et al., 2006; Baggett et al., 2014). Once the oysters reach a given size (perhaps 25-40 mm shell height) or have a thick enough shell, they can be added to reefs on the shell or if larger, seeded directly onto newly constructed reefs (Figures 16b, d). Connections among oyster reefs and regions (Eggleston, 1999; Eggleston et al., 1999; Mroch et al., 2012; Puckett and Eggleston, 2012) are also key to

Bivalve Molluscs, Figure 15 (**a**) Overharvested fringing intertidal shoreline with clusters of *Crassostrea virginica* at fringe of the *Spartina alterniflora* marsh only (cf. Figures 7b & 8b). Lower down on the bank is mostly loose shell, with intact live oyster clusters closer to the marsh, SC, USA (Source: Loren Coen). (**b**) Restored leased shoreline after shell ("cultch") was planted and allowed to recruit with natural oysters ("spat") after several years without harvesting, SC, USA (Note the oyster clusters with numerous vertical oyster "blades") (Source: SCDNR, Charleston, SC).

future restoration success (Lipcius et al., 2008; Lipcius and Ralph, 2011).

Critical for successful restoration efforts are clear goals, related metrics, and success criteria and designs (Weinstein et al., 1997) that are rigorous with adequate monitoring (Coen and Luckenbach, 2000; Coen et al., 2004; NRC, 2010; Kennedy et al., 2011; Baggett et al., 2014; Powers and Boyer 2014). Monitoring also allows for adaptive management of the restoration process in the event efforts also beyond the initial restoration activities are required (Coen et al., 2004; Kennedy et al., 2011; Baggett et al., 2014). Past efforts also suggest that at least four or more years are required to begin to assess long-term success (reviewed in Baggett et al., 2014).

Shell budgets for subtidal oyster reefs in the northeastern USA have been calculated and used to assess reef shell trajectories and the likelihood of long-term restoration success (Powell et al., 2006, 2012; Waldbusser et al., 2013). Intertidal evaluations of natural oyster reef changes and restoration success metrics can be more easily assessed using a number of approaches (Grizzle et al., 2002; Coen et al., 2004, 2011b; Powers et al., 2009; Baggett et al., 2014). For a large number of restoration footprints in North Carolina, Powers et al. (2009) determined that intertidal success was much greater than that observed for subtidal restoration efforts, but this finding may be confounded by a number of potential methodological problems. More work needs to be done with regard to success of small to large footprints (reviewed in Kennedy et al., 2011; Baggett et al., 2014). The large-scale 2009 American Recovery and Reinvestment Act (ARRA) for oyster reef-related projects across the Gulf of Mexico and eastern USA may provide some of these answers.

One significant result of earlier restoration efforts is that for most subtidal restoration, where dissolved oxygen is a major problem (Baker and Mann, 1992;

Bivalve Molluscs, Figure 16 (**a**–**d**) In cases where recruitment potential ("larval supply") is low or where one needs to jumpstart reefs during enhancement or restoration efforts, "spat on shell" (SOS) or seed oyster additions may be used. (**a**) Shell in tanks with oyster larvae added to recruit in the hatchery (Source: R. Grizzle); (**b**) trays of SOS ready to deploy the field in Soundview Park, Bronx, N.Y. as part of a restoration effort lead by Hudson River Foundation, see http://www.hudsonriver.org/?x=orrp (Source: Rocking the Boat, NY, USA); (**c**) natural oyster "spat" collected on Atlantic surf clam (*Spisula solida*) shell in lower Delaware Bay, USA. Spat collectors (shell in bags) were deployed as part of Rutgers University's (New Jersey, USA) community-based oyster restoration program (or PORTS: Promoting Oyster Restoration Through Schools; Source: Lisa Calvo, Rutgers University, NJ, USA); and (**d**) larger "single" 6-month-old oysters are often used to seed reefs (Source: J. Gatling, Kiwanis Club of Suburban Norfolk, VA, USA).

Diaz and Rosenberg, 1995; Lenihan and Peterson, 1998; Lenihan, 1999; Breitburg et al., 2009), higher-relief reefs prove to be more successful than low-relief or no-relief reefs (discussed in Coen and Luckenbach, 2000) in the Gulf of Mexico (e.g., Gregalis et al., 2009), the southeastern USA (e.g., Lenihan and Peterson, 1998; Lenihan, 1999), as well as the mid-Atlantic USA (e.g., Luckenbach et al., 1999; Woods et al., 2005; Schulte et al., 2009).

Mapping

In many areas, major efforts have taken place with new imagery and related mapping (Figure 7, ASMFC, 2007; SCDNR, 2008; Kennedy et al., 2011; La Peyre et al., 2014a) or will be underway (e.g., RESTORE funding for the Gulf of Mexico) to assess the current status and eventually trends for triaging these recovery efforts that require their mapping (Grizzle et al., 2005, 2008b; Powers et al. 2010) for later detection and, if possible, storage in a GIS geodatabase (see http://www.dnr.sc.gov/GIS/descoysterbed.html; Gambordella et al., 2007; SCDNR, 2008; Ross and Luckenbach, 2009, http://www.oyster-restoration.org/oyster-restoration-research-reports/).

Aquaculture

Aquaculture is playing an ever increasing role in the enhancement or restoration of native and nonnative bivalves and more generally molluscs in North America (see Figure 17) (Manzi and Castagna, 1989; Dumbauld et al., 2009; Shumway, 2011) and of late as a potential tool for other nonconsumptive "ecosystem services" for

Bivalve Molluscs, Figure 17 (**a**) Grow-out (predator-exclusion) cages with small oysters held in the water column in shrimp ponds in South Carolina, USA (Source: Bill Cox, Island Fresh Seafood, Meggett, South Carolina, USA). (**b**) Oyster farming on the west coast of USA (Washington, USA), where significant areas are often leased for growing native and nonnative molluscan shellfish species such as oysters (Source: Bill Dewey, Taylor Shellfish, WA, USA).

shellfish worldwide (French McCay et al., 2003; DeAlteris et al., 2004; Beck et al., 2009, 2011; Coen et al., 2007; Coen et al., 2011a; Grabowski and Peterson, 2007; Brumbaugh and Coen, 2009; NRC, 2010; Allison et al., 2011; Powers and Boyer, 2014). Additionally, there are many parallels in the services rendered by farmed and natural reef restoration approaches (e.g., Dumbauld et al., 2009; Coen et al., 2011a), especially since bivalve aquaculture is unique in many ways from other cultured species' approaches in that it requires exceptional water quality for field grow-out (Figure 17, Leonard and Macfarlane, 2011). The shellfish aquaculture industry has helped to improve water quality standards in areas they utilize (e.g., waste water treatment or septic system upgrades), and some have suggested that mussel aquaculture may provide a mechanism for reducing the eutrophication impacts (reviewed in Lindahl, 2011). However, not all of the impacts are strictly positive (Simenstad and Fresh, 1995; Dumbauld et al., 2009; NRC, 2010; Coen et al. 2011).

River diversions

In many estuaries, large-scale diversions and rediversion ("reengineering") of rivers and also seasonal releases or reserves of freshwater (e.g., Louisiana, South Carolina, Texas, Florida, USA) have led to major controversies and related impacts on oyster resources, as well as many other habitats in the overall landscape (Wilber, 1992; Burrell, 2003; La Peyre et al., 2009, 2013; Volety et al., 2009; Pollack et al., 2011). For example, in the ever so important northern estuaries of the Everglades (the Caloosahatchee, Loxahatchee, Lake Worth Lagoon, and St. Lucie , Florida, USA), seasonal wet/dry rainfall variability and related managed pulses or the absence of freshwater can either raise or lower salinities and other environmental variables increasing predators and parasites (when releases are low) or killing estuarine organisms that cannot relocate (e.g., SAV, clams, and reef-building oysters) given the extended periods of these man-made conditions (Tolley et al., 2005; Volety et al., 2009; Volety, 2013). Climate change (including pH and CO_2 levels), diseases, and sea level rise will cause even greater problems in the future (Lafferty et al., 2004; Allison et al., 2011; Levinton et al., 2011; Waldbusser et al., 2013; Burge et al., 2014; Waldbusser and Salisbury, 2014). Enhancement and restoration efforts will play key roles in the future (Blignaut et al., 2013; Powers and Boyer, 2014). The use of shellfish, especially bivalves for nutrient assimilation in estuaries, may also play an increasing role in the future

(Higgins et al., 2011, 2013; Levinton et al., 2011; Shumway, 2011; Piehler and Smyth, 2011; Kellogg et al., 2013; Smyth et al., 2013).

Summary

Bivalves, especially reef-forming species (see DeAlteris, 1988, and Figure 1 in Waldbusser et al., 2013), are important habitat formers in many estuaries worldwide (Kirby, 2004; Beck et al., 2009). Bivalve populations (e.g., mussels) often have positive synergies with other habitats such as sea grasses (Williams and Heck, 2001; Coen et al., 2011a). Similarly, some oyster species (e.g., *Crassostrea gigas*), through direct and accidental introductions, are having significant negative impacts on many native species (Europe, Smaal et al., 2005; Nehls and Büttger, 2007; Kochmann et al., 2008). Impacting one habitat can often impact another in various ways. Because of their numerous ecosystem services, they are in many places being enhanced or restored from current often depauperate levels. A major effort in assessing their current status and eventually trends for triaging these recovery efforts (e.g., in the Gulf of Mexico, post-Deepwater Horizon) requires that habitats be mapped in advance and put into a GIS geodatabase (SCDNR, 2008; see http://www.dnr.sc.gov/GIS/descoysterbed.html). Approaches for their population assessment entails consistent approaches and good designs for monitoring natural and recovering populations. The importance of population connectivity (metapopulations) needs to also be considered for restoration efforts over larger spatial scales (Lipcius et al. 2008, 2009; Schulte et al., 2009). Goals and related success criteria need to be developed whether they are intertidal, shallow, subtidal, or in deeper estuaries and surrounding waters (see http://www.oyster-restoration.org/). Climate change, shoreline erosion (and related fringing habitat loss), changes in native and nonnative (introduced) diseases, competitors, and predator introductions will impact estuaries and the native and cultured bivalves in these systems. Sea level rise, increased hypoxic zones, and other challenges will create habitat winners and losers in estuaries. Oyster reefs are potentially one of the nine important nearshore habitats that will protect coastal communities and infrastructure (Arkema et al., 2013; Grizzle and Coen, 2013). Aquaculture will have an increasing role in bivalve sustainability (Beck et al., 2009; Brumbaugh and Coen, 2009; Dumbauld et al., 2009; NRC, 2010; Shumway, 2011).

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Cross-references

Bivalve Aquaculture
Coastal Erosion Control
Estuarine Habitat Restoration
Habitat Loss
Intertidal Zonation
Introduced Species
Invasive Species
Oyster Reef
Predator–Prey Relationships
Salt Marsh Accretion
Sea-Level Change and Coastal Wetlands
Shell Beds
Shellfish Production
Shore Protection
Shoreline Changes
Thermal Biology

BLUE CARBON

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Synonyms

Coastal carbon; Vegetated coastal carbon sinks

Definition

The term “blue carbon” refers to the proportion of “green” or biological carbon that is found in the oceans of the world (Nelleman et al., 2009). Three main types of coastal ecosystems contain the majority of this blue carbon. Mangroves are a type of tidal, forested wetland found in the tropics and subtropics. Tidal marshes are tidal wetlands dominated by emergent vegetation including grasses, sedges, and reeds. Seagrass beds or meadows are ecosystems along the coasts from the arctic to the tropics containing submerged aquatic vegetation, which resembles terrestrial grasslands. Because these three types of ecosystems store the majority of this carbon, “blue carbon” has become synonymous with coastal carbon. Altogether, mangroves, tidal marshes, and seagrass beds cover roughly 49 million hectares in area (Pendleton et al., 2012) and account for the burial of approximately 114–131 Tg (1 Tg = 1×10^{12} g) C/year (Nelleman et al., 2009). Within these ecosystems, the majority of the blue carbon is stored in soils and sediments; however, in mangrove ecosystems and tidal freshwater swamps, a good proportion of carbon may also be stored in trees.

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Cross-references

Carbon Sequestration
Mangroves
Saltmarshes

BLUE CRABS

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Synonyms

Atlantic blue crabs; Blue claw crabs

Definition

Blue crabs, *Callinectes sapidus*, are an ecologically and economically important crustacean species in estuaries along the east coast of the United States. Their entire range is from Massachusetts to Argentina (Millikin and Williams, 1980).

Summary

Blue crabs are considered estuarine residents with all life history stages, except for the larval stages, occurring in estuarine waters. Adult females release larvae into the water column near the mouths of estuaries (Millikin and Williams, 1980). Larvae are carried offshore where at least 30 days are required to go through seven zoeal stages (Millikin and Williams, 1980; Epifanio, 2007). As a result, blue crab larvae represent one trophic link between estuarine and oceanic food webs. The final planktonic stage (megalopa) returns to the estuary, via wind-driven currents and tides (Epifanio, 2007), where they metamorphose to form the first juvenile stage (<5 mm carapace width) and become benthic. These juveniles grow rapidly, molting on average every 3–4 weeks depending on water temperature (Smith and Chang, 2007), making them important prey for a variety of fish and birds, but they are also important predators on other small invertebrates (Lipcius et al., 2007). Therefore, young juveniles (<20 mm carapace width) occupy shallow, structured habitats that also contain food sources including seagrass beds, macroalgae, and oyster reefs (Lipcius et al., 2007). During the 12–18 months required to reach sexual maturity (at 90–100 mm carapace width), habitat use expands based on size and density-dependent factors (Hines, 2007; Lipcius et al., 2007). Smaller juveniles (20–30 mm carapace width) move to alternative nursery habitats including marsh creeks and marsh-fringed mud flats (Lipcius et al., 2007). Larger juveniles (>20 mm carapace width) begin venturing into unstructured habitats and, as they grow, inhabit deeper areas where they continue to be important predators but are prey to fewer organisms (Hines, 2007).

Unlike females that exhibit a final molt to reach maturity, adult males grow throughout their lives (reaching sizes of >200 mm carapace width), molting every 30–40 days depending on temperature, and they typically return to more protective habitats during molting, as they are particularly vulnerable to predators. As a result of ontogenetic shifts in habitat as well as movement into lower salinity areas, blue crabs can be found in a wide array of habitats, throughout the estuarine-to-ocean salinity gradient (e.g., 5–35 ppt) (Hines, 2007; Lipcius et al., 2007). Because they represent both predator (contrary to popular belief, they are not scavengers) and prey in these habitats, blue crabs are a critical component of the estuarine food web both within and between estuarine habitats.

Blue crabs have been an important food item for humans since the early 1700s and have supported a commercial and recreational fishery since the 1800s (Kennedy et al., 2007). As a result, blue crabs are part of the historic, economic, and social fabric of communities along the Atlantic and Gulf coasts of the United States. Blue crabs are consumed as “hard crabs” (crabs with a hard carapace, typically in the intermolt stage) and as “soft crabs” (crabs with a soft carapace as a result of recent molting) (Kennedy et al., 2007). The predominant fishing techniques vary with the season and with the sex of the harvested crabs. During the warmer seasons (late spring-late fall), blue crabs are typically harvested with a trap or “pot,” and the catch is predominantly males. During the winter, particularly in the mid-Atlantic region, blue crabs are harvested by a dredge, and the catch is predominantly females.

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Cross-references

Soldier Crabs (Mictyridae)

BULKHEADS

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Synonyms

Retaining walls; Revetments; Seawalls

Definition

Vertical structures or partitions that hold or prevent soil from sliding seaward and reduce land erosion. A secondary purpose of these structures is to provide protection to the upland from light-to-moderate wave action (CHL, 2013).

Bulkheads protect bluffs and cliffs by retaining soil from eroding at the toe, thereby increasing stability. Bulkheads may cause increased erosion immediately seaward and adjacent to the structure (flanking) due to wave reflection, and they offer no protection to adjacent areas. Bulkheads may be cantilevers, anchored (e.g., sheet pile), or gravity structures (e.g., stone) (USACE, 1981; USACE, 1984; USACE, 2002).

Cantilever bulkheads require adequate ground embedment to retain soil and prevent overturning and are typically used where lower structures are needed. Scour at the toe of the structure can effectively reduce the embedment length and cause failure (USACE, 1981).

Anchored or tie-backed bulkheads require adequate embedment (less than cantilever bulkheads), gain additional support from anchors embedded on the landward side or from structural piles placed at a batter on the seaward side, and are usually used where higher structures are needed. Anchored bulkheads tend to be less susceptible to toe scour; however, they require corrosion protection at the connectors (USACE, 1981).

Gravity bulkheads require strong foundation soils to adequately support their weight (e.g., gabion baskets and concrete blocks), do not sufficiently penetrate the ground to develop reliable soil resistance, and are appropriate where subsurface conditions hinder pile penetration. Gravity bulkheads are typically low-height structures, depend on shear resistance at the base of the bulkhead to support the applied loads, and cannot prevent rotational slides (USACE, 1981).

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Cross-references

Revetments

C

CARBON SEQUESTRATION

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Synonyms

Carbon capture; Carbon storage

Definition

Carbon sequestration is the process by which atmospheric carbon dioxide (CO_2), the most important greenhouse gas, is removed from the atmosphere and stored in the ocean, on the land surface, or in geological formations (Sundquist et al., 2008). Reservoirs that store carbon over long periods of time are called "carbon sinks." The process of carbon sequestration can occur naturally by plants via photosynthesis with subsequent storage of carbon in biomass (leaves, roots, and stems/trunks of plants) and soils. Carbon can also be sequestered by separating and capturing CO_2 emitted by industrial processes and transporting it to deep underground geological formations for permanent storage (Lal, 2008). Carbon sequestration is reported as a rate of carbon (C) storage in units of mass per time such as teragrams ($Tg = 1 \times 10^{12}$ g) C/year.

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Cross-references

Saltmarshes

CHENIERS AND REGRESSIVE BEDFORMS

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Synonyms

Chenier plain; Sandy beach ridge

Definition

Chenier is a term devoted exclusively to linear sandy coastal ridges separated from the shoreline by muddy deposits (mudflats or marshes). The base of the cheniers should be a horizontal surface where a clear difference exists between the mud and the coarse-grained deposits. The seaward sides of the cheniers are regular and straight, while the landward side is irregular (Reineck and Singh, 1980). Cheniers do not occur alone, but in groups, with narrow spacing between individual ridges at their ends. Moreover, this arrangement can cause streams and small rivers to be deflected, creating angular or parallel drainage patterns, a feature that was noted in the Suriname chenier plain by Augustinus (1980).

Introduction

Cheniers are deposits originating from high-energy reworking of muddy coastal plains. Beach-ridge and chenier plains can be confused when it is difficult to discern if the plains are dominated by sand and mud.

M.J. Kennish (ed.), *Encyclopedia of Estuaries*, DOI 10.1007/978-94-017-8801-4,

Cheniers were originally defined as shallow-based, sandy beach ridges resting on clay along a marshy or swampy, seaward facing, tidal shore, with other beach ridges stranded in a marsh behind, forming a belted marsh-and-ridge plain (Price, 1954, 1955), and usually enriched in up to pebble-size shelly material (Otvos, 2000). In other words, beach-ridge plains have local names in Louisiana (cheniers) and Suriname (ritsen). The term chenier was given to the ridges with oaks growing on them, which are called "chênes" in Southwest Louisiana. The chenier plain comprised of the vegetated marshes, water bodies (including lakes, streams, and tidal inlets), and beach ridges (Byrne et al., 1959). However, these ridges possess certain characteristics. They are less than 3 m high, 30–50 m wide, and no more than 5 m thick. The growth of these cheniers was clearly related to the mud delivered by the Atchafalaya River (Wells and Kemp, 1981).

The link between cheniers and fluvial input has been applied to other deltas or river estuaries. Since the origin of the term, they have been reported at the deltas of the Mississippi, Amazon, Orinoco, Po, and Rhone Rivers (Price, 1954). Wells and Coleman (1977) extended their studies of the suspended sediment transported from the Amazon and Orinoco deltas to the chenier plains of Suriname (Wells and Coleman, 1981). Their model related chenier growth to the number of days when the tide exceeded a certain level, the increase in sediment concentration, the decrease in sediment compaction, and the root density of mangroves at the coastline (Wells and Coleman, 1981).

In recent times, beach ridges are being used as a common term for coastal features originating from several processes: (1) swash action, (2) settling lag, (3) eolian action, (4) and storm surges (Tanner, 1995). They are also related to the episodic input of sediment, either of fluvial (Anthony, 1995) or volcanic origin (Nieuwenhuyse and Kroonenberg, 1994). They can be composed of sand, gravel, or shells (Reineck and Singh, 1980).

Sea-level trend

Although they were originally defined for low-lying deltaic plains affected by rising sea levels, as along the Mississippi River, cheniers are more abundant on plains of regressive coasts subject to periodic or episodic high-energy levels. The mid-Holocene was the time when the alluvial plains became stable (Xiqing, 1996). This occurred around the same time as the mid-Holocene sea-level maximum, which varied from less than 1 m to around 4 m above the current level in the Southern Hemisphere (Isla, 1989). However, this highstand was not uniform or stable. In the Northern Hemisphere, sea level is still rising, while in the Southern Hemisphere, it has been dropping slightly. The differences of 2–3 m in the last 5,000 years have produced significant changes in bedforms developing in estuaries. Morphodynamic models are mostly biased toward transgressive coasts (Tanner, 1995; Hesp and Short, 1999). Although progradational barriers are assumed to have a higher potential of preservation on regressive (falling sea level) coasts, not many models have been proposed for these coasts (Roy et al., 1995). Sediment availability controls the facial relationships either in transgressive or regressive sequences (Davis and Clifton, 1987; Isla, 1998).

Location

Cheniers have been observed in every estuary subject to episodic processes or where the availability of coarse material is episodic. They have been found in every continent, with the exception of Antarctica, from low to high latitudes (Figure 1). Therefore, climate only affects its composition and the type of plant communities (e.g., mangroves or salt marshes, in low or mid-high latitudes, respectively).

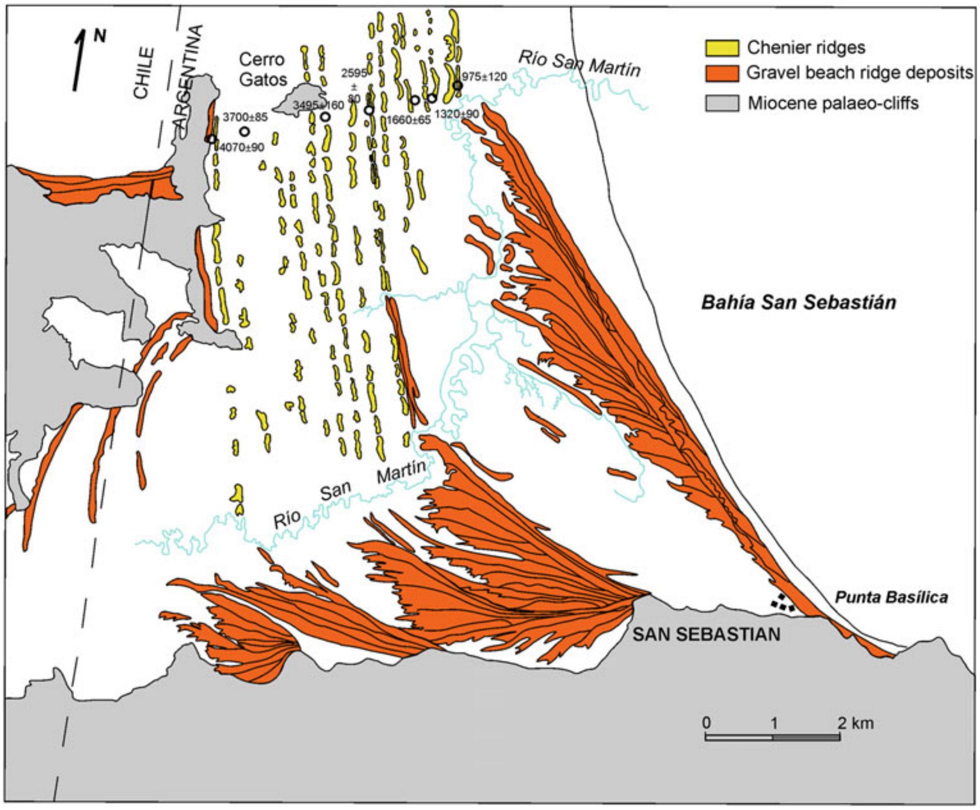
Origin and composition

As their origin implies high energy, cheniers are composed of different materials. Sandy cheniers are quite common, although they can also be composed of shells and gravel. Cheniers have been divided in medium- to coarse-sandy cheniers and fine-sandy cheniers, each one with different accumulation processes, depending on the mode of sandy supply.

Chenier ridges are formed by the interplay between washover and beach drift processes. Sand in low- to medium-energy coasts is effectively transported by constructive wave action. The grains are suspended by the turbulence of the breaking waves and transported by beach drift. During the rising tide, the sediment is stirred at the seaside of a developing chenier by approaching breakers. If the crest is low, sand will be washed over it and deposited on the lee side, causing the chenier to migrate gradually landward. Once the crest is high enough, beach drift becomes the dominant process, and the chenier begins to extend laterally, depending on the direction of the current (Augustinus, 1980).

Another mechanism of chenier formation is the switching of delta lobes (Otvos and Price, 1979; Penland and Suter, 1989). This is particularly evident in the variations of the Huang He (Yellow) River outlet to the North Jiangsu of Bohai Bay (Xitao, 1989; Yan et al., 1989).

Medium- to coarse-sandy cheniers are built up by sand delivered by longshore currents, beach drift, and washover processes. Thus, they form at or just above high-tide level. Sedimentary structures within coarse-textured cheniers include lamination in two different directions – foreslope- and backslope-parallel laminations. The foreslope-parallel lamination occupies a narrow strip on top of the chenier, with the rest of the chenier's body composed of the landward-parallel lamination. Cross-stratification sets may appear intercalated with the parallel laminate depending on the water level landward of the chenier (e.g., in mangroves or salt pans). If the chenier is sufficiently high, small washover deltas develop due to the sudden slowing down of the running water containing



Cheniers and Regressive Bedforms, Figure 1 Location of cheniers (*triangles*) and beach-ridge plains (*circles*).

sediment as it encounters stagnant waters. This process is related to spring tides and ends before the following neap tide (Augustinus, 1980; Augustinus et al., 1989).

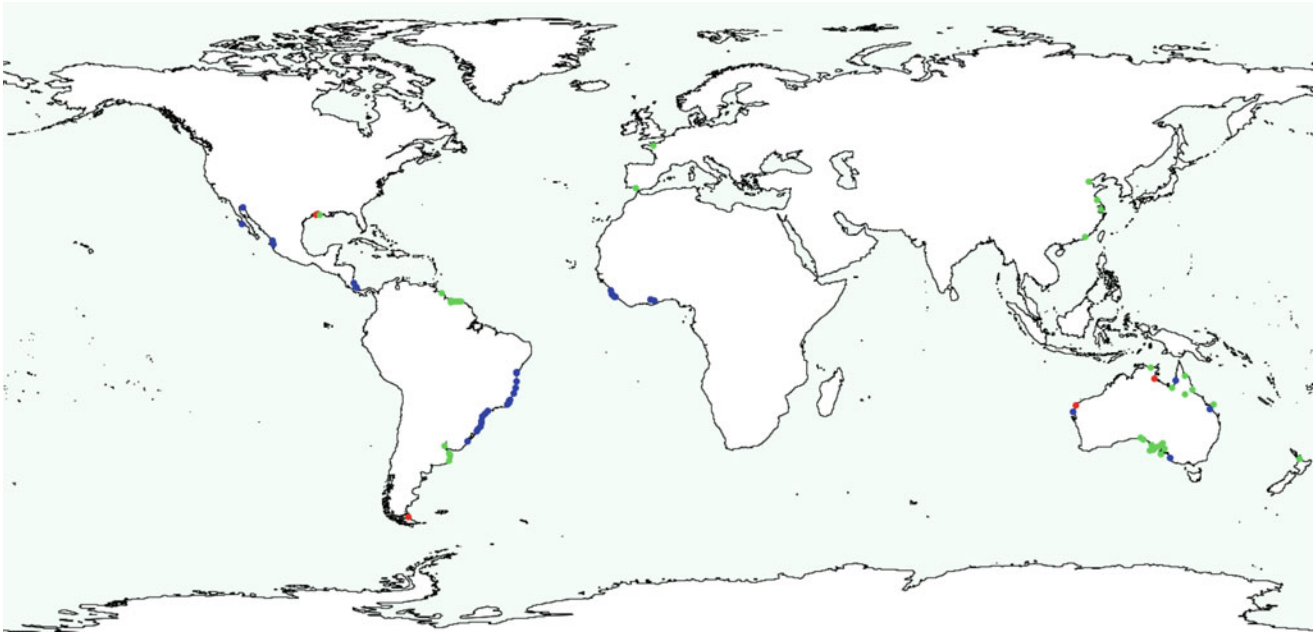
Fine-sandy cheniers originate at the mean low water level, where unsorted mudflat deposits are stirred up by waves and currents. Sand settles on the more tranquil nearshore waters, while the finer sediment is transported farther, forming a longshore bar that moves obliquely landward, gathering volume and height. When the up-current end attaches to the mainland, it appears similar to a spit, with a lagoon-like tidal flat behind it. The main body of the fine-sandy cheniers is composed of a thick bed set of gently seaward-dipping laminate with intercalated thin sets of coarser, steeply landward-dipping shelly layers. The lagoon-like tidal flat is characterized by interlayered sand/mud bedding (Augustinus, 1980; Augustinus et al., 1989).

The transition from beach ridges to cheniers is a matter of sediment availability. In San Sebastian Bay, Tierra del Fuego, a river runs between the boundary of a beach ridge composed dominantly of gravel and a chenier plain composed of shell debris in a sandy matrix and interfingered with mudflats and salt marshes (Isla et al., 1991). Although the beach-ridge plain is assumed to be older, the chenier plain developed between 4,070 and 975 years ago (Figure 2; Vilas et al., 1999). In a similar way, the beach-ridge plain expanded eastward during a period close to the maximum highstand of the headlands of the Rio de la Plata mid-Holocene embayment (Figure 3). About 2,800 years BP, the Parana Delta expanded southeastward, restricting wave processes and increasing the mud availability. This caused a significant change, and a chenier plain that prograded until the deposition of the Parana and Uruguay Rivers generated a single lobe (Figure 3).

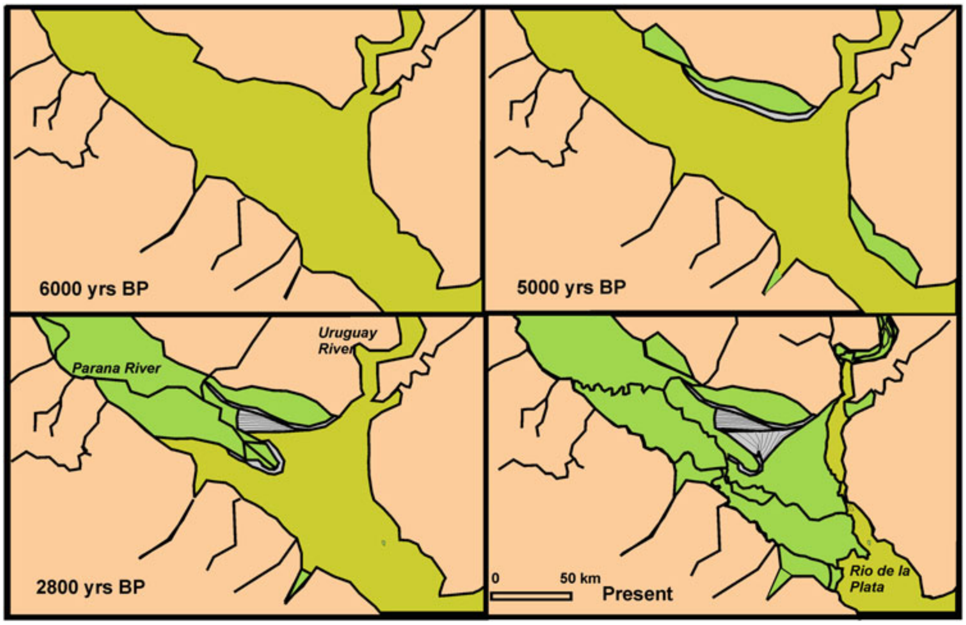
Along the Baja California Peninsula, the tide-dominated coast (macrotidal) has a chenier plain formed by deposition and starvation processes of the Colorado River Delta. On the other hand, the open-ocean coast of Vizcaino Peninsula (mesotidal) beach-ridge plains is associated with high tides and storms (Meldahl, 1995). Applying Ground Penetrating Radar (GPR) techniques, sand-dominated and mud-dominated plains have been differentiated at the Tijucas (meaning “marsh” or “swamp” in the Tupi language) River Inlet, Santa Catarina, Brazil (Buynevich et al., 2005).

Regressive bedforms

More strictly speaking, a chenier plain is composed of at least two subparallel ridges or ridge sets, “sandwiched” between tidal-subtidal mudflats. It represents multiple



Cheniers and Regressive Bedforms, Figure 2 Development of gravel and chenier plains of the San Sebastián Bay, Tierra del Fuego (Modified after Vilas et al., 1999).



Cheniers and Regressive Bedforms, Figure 3 Development of the Paraná-Uruguay composite delta (Modified from Cavalotto et al., 2005).

episodes of ridge and mudflat formation on prograding shore sectors. Mudflat progradation must bracket the ridge: a marsh-covered or barren preexisting mudflat in its rear, and a younger, possibly still active mudflat seaward (Otvos and Price, 1979; Otvos, 2000).

There are two main types of chenier plains depending on the type of coast in which they develop. Bight-coast chenier plains include those forms of the West Louisiana and Guiana coasts. In this setting, a broad indentation of the bight coast contributes to a decrease of wave energy approaching the nearshore and the settling out of muddy deposits. Bayhead chenier plains in turn are formed on bayheads of smaller dimensions than those on bight coasts. Classic bayhead chenier plains include Broad Sound and Burdekin River, Queensland (Cook and Polach, 1973), the Colorado Delta, California Gulf (Otvos and Price, 1979), Firth of Thames, New Zealand (Woodroffe et al., 1983), or at the headlands of the Rio de la Plata (Cavalotto et al., 2005). Other examples are San Sebastian (Vilas et al., 1999) and Samborombon Bays (Bértola, 1994) on the coast of Argentina. Recently, Spanish authors distinguished a third setting for chenier formation: estuarine coasts protected from the open sea by a coastal barrier (Rodríguez-Ramírez and Yáñez-Camacho, 2008). Tentatively, the ridges along the Mar Chiquita lagoon coast fall into this category.

Age

Radiocarbon dates from shells collected from cheniers do not relate directly to ridge accumulation but to the time of the organism's death before ridge formation (Woodroffe, 2002). As the maximum limit of the Holocene transgression occurred ~6,000 years ago, there are differences in the ages of the shells composing beach deposits. Shells sampled in living position from estuarine facies are better indicators of the maximum highstand (Isla, 1989; Cavalotto et al., 2005).

Schofield was the first to relate a chenier plain to the sea-level fall of ~2.6 m in 2,000 years at the North Island of New Zealand. The Miranda plain is composed of cheniers and regressive spits dated between 3,900 and 980 years BP (Schofield, 1960). Recently, the sequence of regressive spits was reanalyzed based on GPR records. Significant differences were recognized based on the architecture of the older ridges (13-6), related to mudflats, and the modern ridges (5-1) deposited on embayed tidal flats (Dougherty and Dickson, 2009).

After the maximum highstand of 5,800–5,500 years BP, a chenier plain developed as sea level dropped at the mesotidal Gulf of Carpentaria, Australia (Rhodes, 1982). At the eastern side of Queensland, another chenier plain developed in Princess Charlotte Bay. From a maximum highstand of ~6,000 years BP, beach ridges and cheniers composed of shells are interfingered (Chappell and Grindrod, 1984). Mangroves colonized cheniers in a prograding shoreline. Managing different chenier plains, significant changes in the progradation were detected over the last 6,000 years at Princess Charlotte Bay, Karumba, and South Alligator River (Woodroffe, 2002). At Southern Australia, another chenier plain composed of shells increased in size due to organic-rich mangrove facies (Short, 1988). This plain expanded to a highstand of 2–3 m above today's sea level ~6,400 years ago. The plain's growth was recorded using organic remains from the transition between *Posidonia* sea grass and sand flats and from these flats to ridges dominated by *Avicennia* mangroves (Belperio et al., 2002).

Cheniers can be used as reliable markers of paleocoastlines, since they only form on stable or slightly retreating coasts. When there is a steady sediment supply along with a gradually falling sea level, the result is an increased rate of mudflat progradation (Chappell and Grindrod, 1984), which prevents chenier formation or lowers their frequency of appearance. Moreover, an increased deposition of muds causes a reduction in shellfish communities, which in turn diminishes shell production and the quantity of the material available to form cheniers. On the northern outer shelf of the East China Sea, submerged chenier ridges were used to map a Pleistocene paleocoastline, with ages between 24,000 and 15,000 years BP. They were found at depths between 150 and 110 m (Xitao, 1989), which reflect the lowstand at the last glacial maximum. The Lelydorp member of the Coropina Formation (Suriname) consists of the deposits of an early Pleistocene chenier plain (Wong et al., 2009).

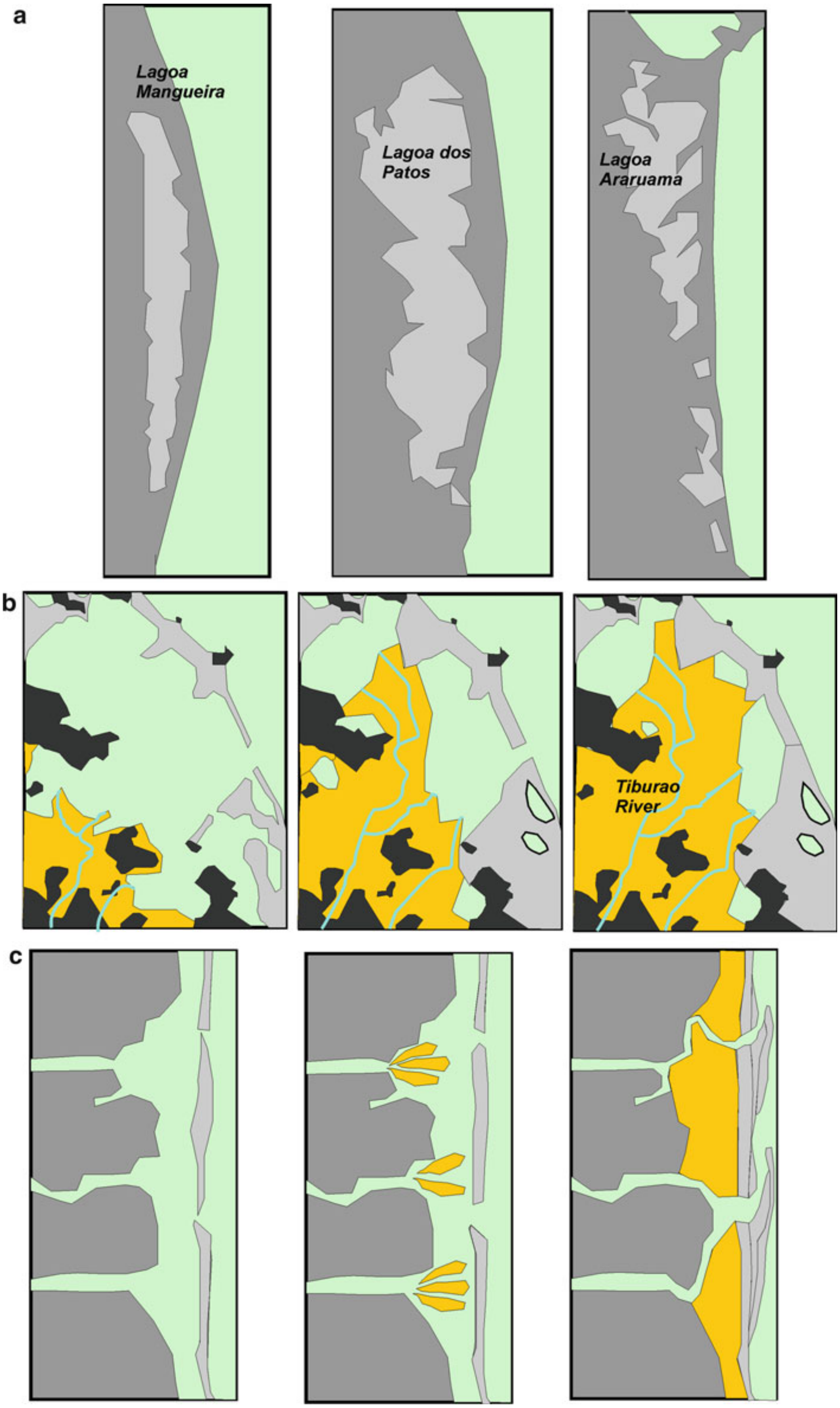
Beach ridges and delta development

Beach-ridge sequences have also been linked to delta development. The so-called wave-dominated deltas of northeastern Brazil grew in relation to beach ridges. During the maximum highstand, coastal lagoons developed and were progressively infilled by intralagoonal deltas (Figure 4; Martin and Dominguez 1994). This regressive model differs from the original view of coastal lagoon development at a coast dominated by longshore drift with a stable sea level (Lucke, 1934; Kumar and Sanders, 1974; Borrego et al., 1993).

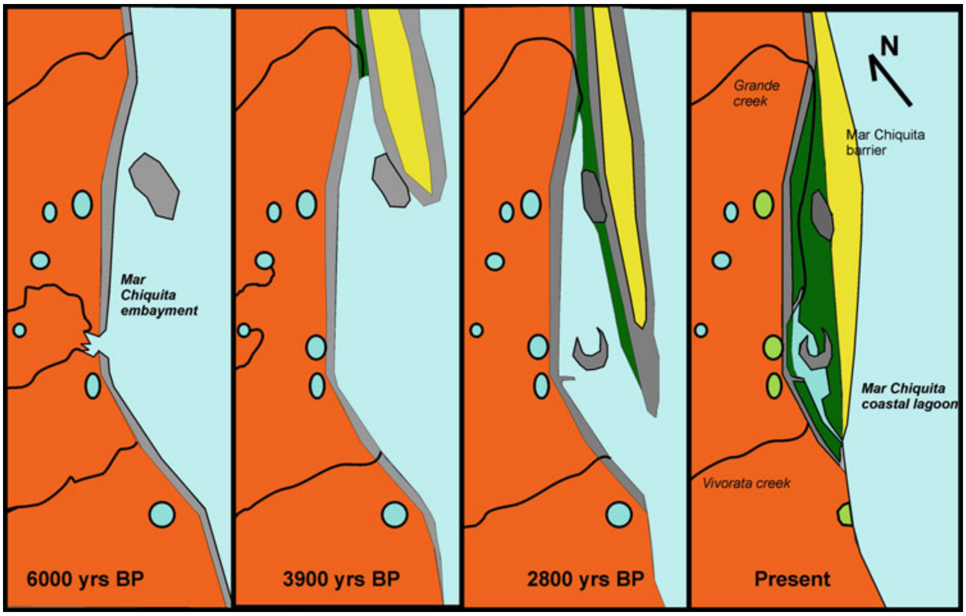
Several deltas were significantly affected in recent years by the construction of river dams or jetty improvements in coastal areas. These changes altered the sediment dynamics of their beach-ridge plains. The Volcán Dam reduced sediment transported to the Orinoco Delta (Warne et al., 2002).

The coast of Nayarit, Western Mexico, provides another good example of a regressive area where coastal lagoons and deltas interact. The region also has a good succession of beach ridges spanning in time from 4,500 years BP to present (Curray et al., 1969). Based on these progradation-aggradation examples, Curray et al. (1969) call attention to non-transgressive models. On transgressive coasts, a barrier is a single ridge transporting onshore. However, other barrier models do exist: (1) a single barrier composed of several ridges that characterize progradation in stable conditions, (2) multiple barrier systems composed of a single ridge each, and (3) multiple barriers composed of several ridges, where each characterizes a regressive coast, related to sea-level fluctuations, climate-triggered net drift changes, or variations in sediment input (deltaic variations).

The beach-ridge plains attached to the progradation of the Doce and Paraiba do Sul developed in response to a steady sea-level fall since the mid-Holocene (Martin et al., 1997). In the evolution of these intralagoonal deltas, the net littoral drift toward the north and the episodic



Cheniers and Regressive Bedforms, Figure 4 Development of Brazilian coastal lagoons. (**a**) Different stages of septation of coastal lagoons (Modified after Isla, 1995). (**b**) Development of a coastal lagoon by septation and deltaic sedimentations (Modified after Nascimento, 2010). Development of deltas filling coastal lagoons (Modified after Dominguez et al., 1987; Martin and Domínguez, 1994).



Cheniers and Regressive Bedforms, Figure 5 Development of the Mar Chiquita coastal lagoon (Modified from Schnack et al., 1982).

effects of waves from the northeast played a significant role (Martin and Suguio, 1992). It is clear that these plains were related to deltas forming within large coastal lagoons until the delta exceeded the body of these bays (Figure 4). Two generations of beach ridges were linked to fluctuations of sea level in a generalized trend of sea-level fall (Martin and Suguio, 1992). Some of these plains were originally prograding cuspate headlands (esporoes or pontales, as they are called in Brazil) which evolved later into regressive spits.

In recent years, different riprap structures have been constructed in order to control the river floods of the Magdalena River (Colombia). The modifications of the discharge channel between 1925 and 1935 produced significant morphological changes (Martínez et al., 1995). The east coast prograded 3.5 km in ~50 years (1943–1990). The two jetties constructed at the outlet (Bocas de Ceniza, 1946–1954) reduced the outlet width to 705 m (Correa et al., 2005; Alvarado Ortega, 2010). On the western coast, the man-made obstruction of longshore drift caused the disappearance of the Sabanilla, Verde, del Medio, and Arena Islands (Martínez et al., 1995).

Processes and mechanisms

Within large estuaries and open coasts, sediment can be distributed by different processes and mechanisms. As a result, several major bedforms occur in these environments that can be distinguished: (1) cuspate spits, (2) regressive spits, (3) cheniers, and (4) washovers.

Cuspate spits are common along elongated coastal lagoons where either wind or tidal currents cause resonance effects, resulting in nodal points where currents counteract and sediment is deposited. This process leads to the septation of coastal lagoons as described by Zenkovitch (1959). It should be stressed that the septation or segmentation is due to estuarine processes (Bird, 2002). The septation of coastal lagoons explains why some cuspate spits are small points, while others are very long and always perpendicular to the main flow (Figure 4).

Some of these cuspate spits can increase in size at a fast rate: the Pontal das Desertas (Lagoa dos Patos, Brazil) prograded about 3,400 m in 57 years toward the southeast (Toldo, 1991). At the region of Laguna (Santa Catarina, Brazil), two beach-ridge plains were compared in their development. The Rio do Meio and Campos Verdes plains (Laguna, Santa Catarina, Brazil) evolved from cuspate spits that divided the mid-Holocene coastal lagoon in the last 5,000 years (Fornari, 2010; Tanaka, 2010). The sediment delivered to this mid-Holocene coastal lagoon by the Tubarão River initially divided the northern lagoons (Santo Antonio and Santa Marta) and afterward the Camacho and Santa Marta lagoons (Nascimento, 2010; Figure 4).

In Mar Chiquita coastal lagoon, there is a cuspate spit 1.9 km long and 300–400 m wide, which shows incipient septation, as it enlarged in a shore-normal direction (Figure 5). The development of the coastal lagoon was influenced by the expansion of a coastal barrier southward (Schnack et al., 1982), while the waterbody was progressively reduced as regressive spits and transverse bedforms composed of shells were forming. Its profile is asymmetric with the higher slope toward the open sea. It is composed primarily by shells of the gastropod *Heleobia* (51–76 %).

Regressive spits occur where there is an interaction between episodic processes (storms) in a regressive trend dominated by longshore drift and a constant siltation rate. Regressive spits occur in sets where it is easy to recognize a different orientation between a set of spits. They can increase in size inside or outside large estuaries and can develop into recurved spits.

The Sao Francisco delta (Brazil) developed within a large coastal lagoon limited by a system of regressive spits (Figure 4; Domínguez, 2009). Fitzgerald and colleagues (1992) described the formation of ridges composed of sand (dominantly) and gravel in response to changes between storm and fair-weather conditions.

Cheniers are caused exclusively by storm effects on a coast subject to wave action or an episodic input of mud. Cheniers should be recognized in vertical sequences where the episodic deposit is overlying estuarine muds.

Washovers consist of coarse deposits on the landward side of a spit or barrier, with sets of landward-dipping lamination or stratification. Their main causes are storm surges, sometimes coinciding with spring tides, which allow the surge to exceed the barrier-spit height. Care should be taken to distinguish between operative washovers, related to present-day storms, and nonoperative washovers that are isolated by processes taking place today. Usually they lie landward of the active washovers.

Summary

Cheniers can occur in different settings where a low-lying plain is subject to periodic high-energy events or drastic fluctuations of sediment availability. The mid-Holocene sea-level fall favors the progradation of chenier plains. Large estuaries, coastal lagoons, and deltas are affected by waves capable of reworking former deposits. Cuspate forelands and spits have been linked to seiches (tidal nodes or wind-generated edge waves). Regressive spits and beach-ridge plains reflect wave effects within estuaries.

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Cross-references

Coastal Barriers
Delta Plain
Sand Ridge
Spit

CLEAN WATER ACT

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Synonyms

Federal Water Pollution Control Act Amendments

Definition

The US Congress enacted the Clean Water Act in 1972 (P.L. 92-500, the 1972 Amendments to the Federal Water Pollution Control Act), which is the principal law dealing with polluting activity in streams, rivers, lakes, and estuaries of the USA (Copeland, 2006). The Clean Water Act established a water quality standards approach for regulating water quality, with the US Environmental Protection Agency responsible for developing national water quality criteria. A waterbody found to be in violation of a water quality standard was to be listed as "impaired" with consideration of the establishment of a total maximum daily load (TMDL) of the pollutant in violation of the standard (Lee et al., 2005).

Description

The Federal Water Pollution Control Act was originally enacted in 1948 and later amended in 1972 as the Clean Water Act (P.L. 92-500). Subsequent amendments were made in 1977 (P.L. 95-217), 1981 (P.L. 97-117), and 1987 (P.L. 100-4). As noted by Copeland (2006), the Clean Water Act consists of two main parts: "regulatory provisions that impose progressively more stringent requirements on industries and cities in order to meet the statutory goal of zero discharge of pollutants, and provisions that authorize federal financial assistance for municipal wastewater treatment construction."

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Cross-references

Anoxia, Hypoxia, and Dead Zones
Eutrophication
Halogenated Hydrocarbons
Nonpoint Source Pollution
Oil Pollution
Polycyclic Aromatic Hydrocarbons
Trace Metals in Estuaries
Water Quality

CLIMATE CHANGE

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Synonyms

Climate variability

Definitions

Climate change (including climate variability) refers to regional or global changes in mean climate state or in patterns of climate variability over decades to millions of years often identified using statistical methods and sometimes referred to as changes in long-term weather conditions (IPCC, 2012). Climate is influenced by changes in continent-ocean configurations due to plate tectonic processes, variations in Earth's orbit, axial tilt and precession, atmospheric greenhouse gas (GHG) concentrations, solar variability, volcanism, internal variability resulting from interactions between the atmosphere, oceans and ice (glaciers, small ice caps, ice sheets, and sea ice), and anthropogenic activities such as greenhouse gas emissions and land use and their effects on carbon cycling.

Introduction

Earth's climate has varied over all timescales due to changes in global energy balance and radiative forcing caused by changes in solar radiation reaching Earth's atmosphere, volcanism, Earth's orbital configuration (precession, tilt, eccentricity), atmospheric greenhouse gas concentrations, and ocean basin-continent distributions. Regional and global climate changes can be amplified or dampened by complex feedback mechanisms involving sea-ice albedo, methane release from permafrost and marine sediments, land surface vegetation cover, ice sheet dynamics, and atmosphere-ocean-land exchange of carbon dioxide. In addition to natural climate changes, there is substantial evidence from instrumental records, climate modeling, and paleoclimate reconstructions that humans have influenced global and regional climate.

Estuaries, inlets, bays, fjords, tidal marshes, and other coastal systems are directly or indirectly affected by climate change. Instrumental records from stream gauges, water quality measurements, and, more recently, satellites provide trends in salinity, temperature, turbidity, dissolved oxygen, and many other parameters that can be linked to regional climate change and variability. Climate, hydrological, and ecosystem modeling studies are another approach to understanding climate impacts. For example, there are growing efforts to project estuarine response to elevated greenhouse gas concentrations (Najjar et al., 2010) some using downscaling methods that link global climate and regional models (Hostetler et al., 2011). Because instrumental records are usually limited to the last few decades, a third approach employs paleoclimatic and paleoecological reconstructions, obtained from geochemical, physical, or biological proxies recovered from sediment cores (Cronin and Walker, 2006; Gooday et al., 2009). Paleo-reconstructions provide direct evidence for past climate impacts and prehistorical baseline conditions for ecosystem restoration, impact assessment, and planning.

Identifying climate impacts on modern estuaries is complicated by multiple environmental stresses from a wide range of local and regional anthropogenic activity such as land use changes associated with urbanization, agriculture, and other activities (Willard and Cronin, 2007; Canuel et al., 2010). These factors can make the attribution of observed changes in estuarine environments to specific causes very challenging.

This chapter summarizes climate impacts on estuaries during the mid-Holocene to late Holocene interglacial period (the last ~7,000 years), which is the period since postglacial sea-level rise stabilized and modern coastal systems took their modern form. Thus, this chapter applies to both prehistorical natural climate variability and climate change since the onset of the Anthropocene, sometimes defined as the period since the industrial revolution beginning ~1750–1800 CE (Common Era) (Gale and Hoare, 2012). Climate impacts can be grouped into five broad, interconnected categories: regional precipitation, sediment processes, temperature (global and regional), biogeochemical processes, and sea-level rise.

Regional precipitation

Climate change has direct impacts on estuaries through its effects on regional rainfall patterns. Seasonal and/or mean annual precipitation in watersheds is often highly correlated with river discharge into estuaries, which in turn affects salinity patterns and circulation. For example, in a partially mixed, microtidal estuary like Chesapeake Bay, river discharge, along with wind and tidal forcing, affects buoyancy-driven circulation, stratification, the development of a pycnocline, and oxygen exchange between upper and deeper layers (Schubel and Pritchard, 1986).

As a consequence, river discharge affects nutrient influx, phytoplankton blooms, dissolved oxygen, water quality, and ecosystem functioning such that excess nutrient loading coupled with greater river discharge has led to estuarine eutrophication on a global scale (Diaz and Rosenberg, 2008; Kemp et al., 2009; Howarth et al., 2011).

Internal modes of climate variability

The term "internal modes of climate variability" is often used to refer to climate changes that are not forced by radiative forcing from GHGs and solar and volcanic activities but rather interactions between the atmosphere, oceans, and ice sheets. The most widely recognized climate patterns are called the El Niño-Southern Oscillation (ENSO), North Atlantic Oscillation (NAO), Pacific Decadal Oscillation (PDO), and Atlantic Multidecadal Oscillation (AMO). Many studies have demonstrated a strong connection between internal modes of climate variability over interannual to multidecadal timescales and estuarine circulation, salinity, and dissolved oxygen (DO). Using a global dataset, Gilbert et al. (2010) could identify a secular pattern of decreasing DO between 1976 and 2000 that was more evident in coastal regions than in the open ocean. However, they also stressed that when interpreting the twentieth century patterns of oxygen concentrations, decadal climate variability can impose large-amplitude oscillations larger than the overall linear trend (see Garcia et al., 2005). Some examples of climate variability impacting regional rainfall, river discharge, estuarine salinity, and, in some cases, nutrient flux include studies of the PDO (Xu et al., 2012), the NAO (Cronin et al., 2005; Prasad et al., 2010), ENSO (Swart et al., 1996; Schmidt et al., 2001; Cronin et al., 2002), and the AMO (Enfield et al., 2001).

There are also well-established links between climate variability and marine biological systems (Mantua et al., 1997, Drinkwater et al., 2003; Pershing et al., 2005; Greene and Pershing, 2007). Cloern et al. (2010) showed that biological communities in San Francisco Bay are sensitive to ocean currents, temperatures, and coastal upwelling connected to PDO variability and North Pacific gyre circulation. Paerl et al. (2013) showed that climate-driven changes in river discharge to North Carolina estuaries altered the composition and biomass of phytoplankton communities. ENSO- and NAO-connected climate variability also influences outbreaks of infectious diseases on a global scale (Lafferty, 2009; Morand et al., 2013) and, in particular, viruses, bacteria, and infectious disease outbreaks in coastal waters (Lipp et al., 2001; Rose et al., 2001).

Two specific aspects of climate that deserve attention are extended droughts or wet periods and extreme events such as tropical cyclones. Evidence from tree-rings, corals, sediments, molluscan isotopes, and speleothems shows that droughts are an inherent part of Holocene climate. Quantitative reconstructions of precipitation show that North America (Cook et al. 2014) and Europe (Büntgen et al., 2010) have experienced decadal, continent-scale droughts over the past millennium. Multiple paleo-reconstructions based on several proxies show that droughts frequently affected mid-Atlantic climate and Chesapeake Bay watershed (Stahle et al., 1998; Cronin et al., 2005, Saenger et al., 2006; Harding et al., 2010). Precipitation changes over centennial timescales also affected coastal systems, such as changes in runoff and productivity in Chilean fjords during the latter part of the Little Ice Age from ~1600 to the 1800s (Rebolledo et al., 2008).

Although specific weather events cannot be directly linked to climate change, there is nonetheless concern that changing climate might increase the frequency and intensity of tropical storms (Nicholls et al., 2007), which can severely impact estuaries. In one of the first intensive studies of hurricane impacts, the June 1972 storm *Agnes* in the eastern United States, there were widespread, long-lasting effects on Chesapeake Bay circulation, salinity, water quality, and ecosystems (Bailey et al., 1975; Davis et al., 1977). Similarly, three hurricanes that hit coastal North Carolina in 1999 caused 50- to 500-year floods, lowered salinity, and enhanced nitrogen loading to Pamlico Sound, which together had multiyear effects on coastal ecosystems (Paerl et al., 2001). Large storms also affect coastal wetlands notably through the impacts of storm surge, wind, and freshwater flushing on wetland soil dynamics and elevation (Cahoon, 2006).

Modeling precipitation changes and impacts

One challenge in estuarine research is predicting future precipitation/streamflow changes due to higher CO_2 concentrations. In the mid-Atlantic region of the eastern United States (Chesapeake Bay, the Delaware Bay, and Hudson River Estuary), impacts on streamflow ranged from a decrease of 40 % to an increase of 30 %, although results varied by season (Najjar et al., 2009). In a study of San Francisco Bay, Knowles and Cayan (2002) found that changes in winter snowpack and reduction in spring runoff would lead to elevated salinity (see Cloern et al., 2011). Future progress in this emerging field will come from linking downscaled climate models with watershed and estuarine hydrodynamic models, especially as improvements are made in predicted precipitation response to future climate change.

Global and regional temperature

Compared to precipitation-driven changes, the impacts of changing temperature may be less obvious in the short term, but nonetheless aquatic temperatures are important in estuarine functioning and ecosystems. Global mean annual and regional ocean temperatures are expected to rise over future decades to centuries due to elevated atmospheric CO_2 concentrations (Najjar et al., 2009), and in theory, this warming might lead to poleward range shifts in temperature-sensitive species (Helmuth et al., 2002; Przeslawski et al., 2012). Moreover, there is indisputable evidence that the world's oceans have been warming for at least the last 50 years (Levitus et al., 2012), and paleoclimate records show that marine species experienced large climate-driven biogeographic range shifts over 10^4–10^7 year timescales. These shifts are best documented in marine sediment records of major microfossil groups (diatoms, dinoflagellates, foraminifera, radiolarian, ostracodes) during glacial-interglacial cycles of the 500,000 years when Earth's mean annual temperature fell ~5 °C during glacial periods (Kucera et al., 2005). In addition to open-ocean sea faunal and floral biogeographic shifts, paleo-records from estuaries and coasts also show Holocene temperature-induced biogeochemical and productivity changes such as the sedimentary record of LIA cooling in Kagoshima Bay, Japan (Kuwae et al., 2007).

In addition to large-scale range shifts, several indirect impacts of rising temperatures deserve mention: reduced sea ice, especially in marginal subarctic seas; coal bleaching; expanded geographic ranges of harmful algal bloom species; and mangrove species expansion among others (Nicholls et al., 2007). Case studies include the Bering-Chukchi Seas (Grebmeier, 2012), the Changjiang River Estuary (Ma et al., 2009), Mediterranean coastal systems (Bensoussan et al., 2010), Narragansett Bay, Rhode Island (Nixon et al., 2009), and the Gulf of Mexico (Bianchi et al., 2013).

Sediment processes

Coastal sedimentary processes influenced by climate include erosion (in the watershed and estuary), transport (in suspension and along river and estuarine bottoms), and deposition in an estuary, bay, or fjord. However, deciphering climate impacts on sedimentation is difficult due to large-scale anthropogenic activities. On the global scale, Syvitski et al. (2005) estimate that humans account for 2.3 ± 0.6 billion metric tons per year but that sediment retention in reservoirs, totaling 100 billion metric tons (bmt) in recent decades, reduces the sediment reaching the world's coasts by 1.4 ± 0.3 bmt per year (see Milliman and Farnsworth, 2011). On a regional scale, Saenger et al. (2008) found that postcolonial agricultural land clearance in the Chesapeake Bay watershed increased sediment accumulation rates by several times, but there were complex leads and lags related to climatic factors.

Nonetheless, preindustrial climate changes are known to affect sediment flux to coastal systems. For example, in subpolar fjords in Svalbard, Szczucinski et al. (2009) found that post-Little Ice Age temperature increase and glacier retreat had large impacts on sediment accumulation.

Within an estuary or bay, sediment affects a variety of factors including turbidity, light penetration, and the distribution of submerged aquatic vegetation (including sea grasses). This applies both to clastic sediment, often referred to as mineral matter, and particulate organic material, much of which is produced by algal productivity fueled by high nutrient concentrations. Sediment also plays an important role in the development of estuarine turbidity maximum zones (ETM, also called turbidity maximum zones, TMZ), a characteristic feature of many estuaries. It has long been known that trapping of suspended material in ETMs can be enhanced by increased vertical stratification due to large freshwater influx (Geyer, 1993). The physics of circulation near these salinity gradients are such that they trap clastic sediment and phytoplankton-derived organic material that has been transported to or resuspended within the ETM, resulting in high nutrient concentrations (Uncles et al., 2006;

Doxaran et al., 2009). As zones of complex salinity variability, nutrient dynamics, planktonic productivity, and fish spawning and growth, ETMs are important estuarine features forced by climate, river discharge, salinity, and sediment transport.

Despite the complexity of processes controlling sediment, land-to-estuary sediment flux, estuaries will continue to be vulnerable to future changes in climate, including the incidence and intensity of extreme storm events.

Biogeochemical processes

In addition to biogeochemical changes related to nutrient and oxygen dynamics discussed above, changes in ocean carbonate chemistry due to the uptake of anthropogenic CO_2 by the world's ocean, often referred to as "ocean acidification" (OA), pose complex, taxon-specific, and still poorly understood impacts on marine life (Hendriks et al., 2010; Wittmann and Pörtner, 2013; Kroeker et al., 2013). It is estimated that mean global ocean pH has been lowered by 0.1 pH units since ~1750 and may decrease by 0.3–0.4 pH units by 2100 (Pelejero et al., 2010). For comparison, glacial-interglacial cycles of the last 400 ka may have experienced changes of between 0.15 and 0.3 pH units. Although anthropogenic driven pH changes cannot be directly compared to natural events due to differing rates and boundary conditions, paleoclimate studies show that over multimillion year timescales, past natural acidification events had large effects on marine organisms (Kump et al., 2009; Pelejero et al., 2010; Hönisch et al., 2012).

Currently, the study of OA impacts on coastal marine organisms is a growing field for corals (Hoegh-Guldberg et al., 2007), molluscs (Talmage and Gobler, 2009; Waldbusser et al., 2011, 2014; Gobler and Talmage, 2013), and other taxonomic groups (Ries et al., 2009; Kroeker et al., 2010). Some case studies suggest that pH has fallen in recent decades in some coastal systems. For example, pH fell from ~8.2 to 7.9 in the last 30 years in Chesapeake Bay (Waldbusser et al., 2011); Feely et al. (2010) estimate that 24–49 % of observed pH lowering in parts of Puget Sound, a deep estuary in the Pacific NW, was due to influx of seasonal upwelled ocean water, that is, global OA, as distinct from in situ remineralization via respiration. Complicating the issue of causality of observed changes in coastal pH, Pelejero et al. (2005) found that pH variation in a southwest Pacific Ocean coral was related to multidecadal climate variability in the Interdecadal Pacific Oscillation. In addition, other factors, such as reduced freshwater influx and higher salinity, may affect estuarine pH.

Sea-level rise

Sea-level rise (SLR) is one of the most challenging yet misunderstood concerns for estuaries and other coastal systems. No fewer than five global and four regional processes influence relative sea level along any particular coast (Cronin, 2012). Global factors include thermosteric ocean expansion (increase in ocean volume, Willis et al., 2010), melting land-based ice from glaciers (increases ocean mass and mean global sea level), melting parts of the Greenland and Antarctic Ice Sheets (increases ocean mass and sea level, Hanna et al., 2013), reservoir storage (decreases mean sea level), and terrestrial water depletion (increases mean sea level, Konikow, 2011). Regional processes (excluding rapid tectonic movement) include glacio-isostatic adjustment (GIA, Peltier and Fairbanks, 2006) due to viscoelastic response of Earth's mantle to melting large ice sheets since the last glacial period ~20 ka (local GIA can also occur due to glacier melting), elastic deformation of Earth's crust due to changes in gravity and rotation (Tamisiea and Mitrovica, 2011), local groundwater withdrawal, and long-term thermal subsidence of the crust (typically minimal).

The contribution of each factor will vary regionally, but nonetheless, from the standpoint of estuaries and other coastal systems, several points deserve emphasis. Global mean sea level has been rising at rate of 3.1 mm $year^{-1}$ over the past few decades (perhaps an acceleration over rates averaged for the last century), mostly due to thermosteric expansion and land ice melting. Some studies suggest that SLR is already affecting large estuaries such as Chesapeake Bay (Hilton et al., 2008; Murphy et al., 2011) and coastal wetlands (Cahoon et al., 2006). In addition, although no consensus exists on future SLR, rates are expected to increase and glacier and ice sheet mass balance loss is likely to dominate SLR the rest of the twenty-first century. Consequently, the modeling study by Hong and Shen (2012) on the impacts of future SLR on Chesapeake Bay is illustrative, finding that primary effects on salinity, stratification, circulation, nutrient retention, and dissolved oxygen varied spatially, seasonally, and interannually. In addition, if as expected, tidal ranges and wave heights increase, severe storms would become an even larger concern in some estuaries (Najjar et al., 2010). Finally, geological records show that in the past, SLR rates reached and at times exceeded ~10–15 mm $year^{-1}$ in the absence of abrupt increase in greenhouse gas forcing. The implication is that, although the many factors that govern coastal ecosystem functioning cannot be oversimplified, the ability of some sensitive systems, notably mangroves, salt marshes, and coral reefs, to "keep up" with SL, that is, to accrete at the same rate of SL rise, remains a major concern.

Summary

Climate changes throughout geological history have influenced estuaries and coastal systems in a variety of ways and over all timescales. Similarly, future climate change will influence estuaries, perhaps at an accelerated rate, notably through effects on salinity and temperature, dissolved oxygen concentrations, nutrient and sediment flux, biogeochemical processes, and coastal ecosystem functioning. Sea-level rise, altered rainfall patterns leading to extreme droughts and wet periods, and biogeochemical changes associated with ocean acidification are among the most important research topics associated with climate that will likely see great progress in the next few years.

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Cross-references

Barrier Island
Estuarine Circulation
Estuarine Geomorphology
Eutrophication
River-Dominated Estuary
Saltmarshes
Sediment Erosion
Sediment Transport
Shoreline Changes
Storm Surges

COASTAL BARRIERS

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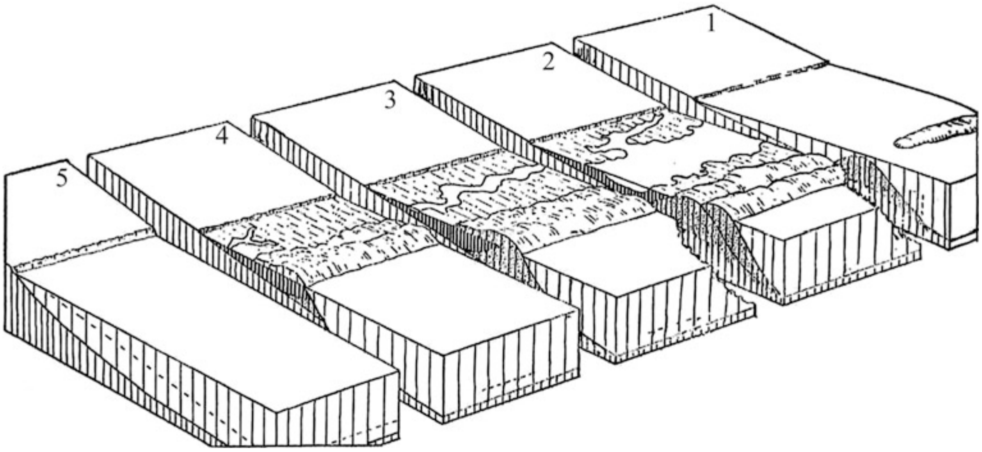
Definition

A "coastal barrier" is a barrier that lies between a sea/lake/lagoon and some landform or feature that is non-coastal or at least more landward than the immediate modern or Holocene coastal landform or group of landforms. It may



Coastal Barriers, Figure 1 Stages in the development and retrogression of a barrier. (After Davis (1912) from Johnson (1919)). The models illustrate some of the barrier types from attached barrier (*5*) to barrier islands (*1*).

be that the next landward feature is another coastal barrier or ancient rocks. In this use of the term, it is merely a barrier between the water and other land or landforms, and it is likely consistent with the original use of the term (Johnson, 1919).

Description

In the American literature, it has been common to refer to a barrier as a barrier island, due to the predominance of these types of barriers on the East Coast of the USA. However, a barrier island is only one type of a suite of types of coastal barriers. Dillenburg and Hesp (2009) state that "a coastal barrier is a shore parallel structure, formed by an accumulation of sand, gravel, shells, and small amounts of organic material due to the action of waves, tides and winds" (p. 1). In some cases, a barrier may not be shore parallel, particularly where spits are forming (Zenkovich, 1967).

Hesp and Short (1999) define a coastal barrier as "a shore-parallel, sub-aerial and sub-aqueous accumulation of detrital sediment formed by waves, tides and aeolian processes. It constitutes a definable coastal landform or sequence of landforms which is clearly separate in age, lithology, and/or form from adjacent, underlying or landward landforms. The barrier may block off or impound drainage from the hinterland, but this is not a prerequisite for definition as a barrier" (p. 308). Coastal barriers may be progradational (building seawards), retrogradational (eroding landwards), or stable (Morton, 1994), and they may be transgressive where sea level is rising and regressive where sea level is falling (Hesp and Short, 1999).

There are a variety of coastal barrier types and many varying terms for these types. On an evolutionary continuum, coastal barriers range from barrier islands (i.e., a barrier separated from the mainland by a lagoon or sea with no connections to the mainland at either end, thus a true island) to attached barriers (i.e., barriers that are attached to the mainland and may merely be a beach or have dune fields which transgress or climb the mainland terrain) (Figure 1). Where the barrier is predominantly a beach and attached to the mainland, it has also been termed a bayhead beach (Johnson, 1919) and mainland beach (Roy et al., 1994). In between these types are various types such as barrier spits (barrier connected at one end to the mainland) and bay barriers (barrier connected at both ends and extending across a bay) (Shepard, 1960; Dillenburg and Hesp, 2009).

The subaerial morphology of coastal barriers may range from a beach and backshore, overwash terrace, overwash fans and nebkha (discrete small dunes), beach and foredune, beach and beach ridge, multiple foredunes (relict foredune plains), beach ridge plains, foredune and blowouts, parabolic dunefields and transgressive dunefields, or combinations of these. All of the dune types may be found on progradational, retrogradational, or stable barriers.

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Cross-references

Backbarrier
Barrier Island

COASTAL BAYS

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Synonyms

Coastal lagoon; Estuary

Definition

Coastal bays are bodies of water of variable size, shape, and morphology formed by the indentation or concavity of the coastline of an ocean or sea. Some investigators include coastal lagoons or enclosed embayments in this definition (Anderson et al., 2010; Glibert et al., 2010). Coastal bays are substantially smaller than a bight (e.g., New York Bight), gulf (e.g., Gulf of Mexico), sea (Sea of Japan), or sound (Long Island Sound).

Description

A range of coastal bays exists worldwide from water bodies largely exposed to the sea to those totally enclosed or nearly totally enclosed by the sea. As stated by Oertel (2005), "headland shores are often seaward of an inundated shore forming an irregular coastline." These headlands usually consist of harder rocks or consolidated sediments more resistant to erosion than those underlying the inundated shore. As a result they form promontories, with the softer substrate, which erodes more rapidly, forming the concavity of the shoreline. This process can lead to the formation of coastal bays.

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Cross-references

Coastal Lagoons
River-Dominated Estuary
Tectonic Eustasy

COASTAL CLIFFS

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Synonyms

Fossil cliffs; Notches

Definition

Coastal cliffs are steep erosional slopes bordering the sea or estuary, in rocky as well as in sedimentary environments. They are common features all over the world (Trenshaile, 1987; Sunamura, 1992; Griggs and Trenshaile, 1994).

Description

In coastal erosive environments, the shore often takes the form of a steep slope or cliff (smaller forms ~1 m in size are termed notches). When formed in rocks, the cliff may terminate in the sea or estuary. In most cases, there is a small active shore just beneath the cliff. In uplifted areas, fossil cliffs (or dead cliffs) may occur. They reflect previous sea-level positions. Along subsiding coasts, one may find submarine drawn cliffs.

Cliff erosion usually feeds lateral accumulation of beach material by long-shore drift. The position with respect to mean sea level of the breaking point between an actively wave-washed shore and a steeply rising cliff foot depends on a number of different dynamic factors (which may change over time).

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Cross-references

Coastal Landforms
Uplifted Coasts

COASTAL EROSION CONTROL

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Definitions

Coastal erosion is a natural or anthropogenic process in which sediment is worn away from the shoreline and seafloor due to natural and anthropogenic factors, such as storms, boat wakes, tidal currents, and rising sea levels.

Erosion control refers to erosion mitigation techniques based on soft and hard structural shoreline stabilization methods and nonstructural measures.

Hard structural stabilization refers to shoreline erosion control approaches based on the construction of man-made structures, such as seawalls, breakwaters, and groins.

Soft structural stabilization refers to shoreline erosion mitigation and control measures based on soft methods, namely, sand, pebble, or gravel fill, such as beach nourishment.

Nonstructural measures refer to any coastal erosion control strategy that does not involve man-made construction or other physical measures, but is based on good practices, policies, and education aimed at reducing anthropogenic and natural impacts, such as land use restriction and zoning.

Introduction

Estuaries are transitional zones where marine and riverine environments meet, i.e., where freshwater from a river mixes with saltwater from the sea. Here, many habitats, species, and ecological communities exist, and their ecosystem and naturalistic values are widely recognized. Along coasts, river banks, and nearshore profiles are governed by sediment transport equilibrium (i.e., erosion and deposition phenomena). Alteration of this equilibrium may result in shoreward recession, leading to land loss. Since estuarine areas are often used for tourism and recreational purposes, this leads to financial loss as well as ecological community and biodiversity impairment. State and local authorities thus aim to protect estuarine areas and to mitigate erosion. A review of the causes of coastal erosion and an evaluation of the erosion processes are provided below, together with an analysis of erosion control measures.

Causes of coastal erosion

There are many causes of coastal erosion processes attributable to natural and anthropogenic factors. A classification can be based on the temporal scale of such factors, distinguishing between short- and long-term events. Natural processes consist of short-term events that are generally the result of storms and river floods (i.e., high-energy content events), while long-term events relate to sea-level rise (Pranzini and Rossi, 1995; Khalil, 1997), tidal cycles, tectonic events, coastal subsidence (Khalil, 1997), climate change, river regimes, and discharge flux (Medina and Lopez, 1997).

In regard to anthropogenic factors, sediment loss is generally due to medium- and long-term events, such as decreasing sediment supply to coastal physiographic units (Simeoni et al., 1997; Eronat, 1999; Loizidou and Iacovou, 1999), deforestation in coastal and riverine watersheds (Eronat, 1999), non-sustainable man-made coastal structures and urban development (Fathallah and Gueddari, 2001; Rakha and Abul-Azm, 2001), flow regime and engineering structure changes, and riverbed sand and gravel extraction (PAP/RAC, 2000).

Erosion processes

Suspension and bed load transport can be distinguished (Fredsøe and Deigard, 1994), the latter mainly causing the loss of grain size material on the seabed and affecting long-term and short-term shoreline evolution. Such phenomena are primarily modeled by physical and mathematical and generally consider hydrodynamic and morphological factors. Hydrodynamic models are based on the classical equations of motion, with vertical averaged velocities (two-dimension models), and wave propagation, refraction, diffraction, and breaking. They evaluate velocity flow fields under generic forcing (Figure 1).

Morphological models consider seabed characteristics and evaluate bed load and shoreline evolution, for both long-term and short-term conditions (Figure 2).

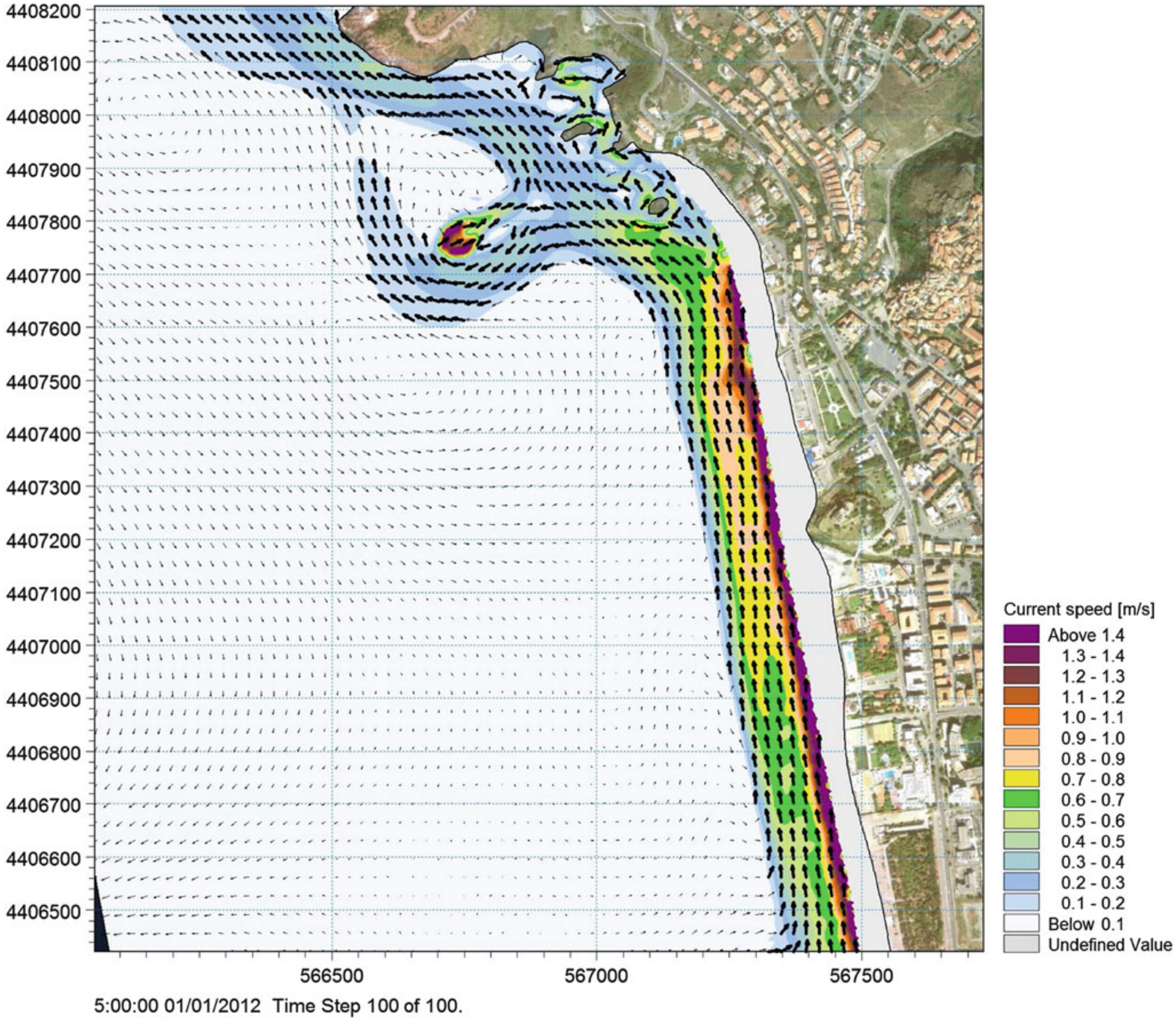
Physical models are based on two-dimensional and three-dimensional laboratory scale similarity models that use channels or large basins (Figure 3) to study both bed load processes and evaluate the effectiveness of the structural design.

Finally, it should be pointed out that both physical and mathematical models require a detailed knowledge of seabed morphology, waves and currents (i.e., river or tidal currents or wave-generated currents), bathymetric surveys, and sediment characterization (pebble or sand beaches, sediment grain size, and erodibilty changes).

Coastal erosion risk mitigation

A coastal erosion control project should be developed with reference to coastal cells (Eurosion, 2004), which are lengths of coastlines in which a complete sediment balance can be identified. Spatial and temporal erosion phenomena scales should also be identified. Acute and structural erosion can be distinguished. Acute erosion is connected to waves, wind, and tidal action and typically covers temporal and spatial scales up to 1 month and from about 1 m to 100 km, respectively. On the contrary, structural erosion involves larger temporal and spatial scales, from about 1 month to 100 years and from 1 to 1,000 km, respectively (Safecoast, 2008).

Depending on spatial scale, coastal erosion control measures can target (1) specific river bank sections and



Coastal Erosion Control, Figure 1 Numerical results of hydrodynamic coastal field at Scalea beach, Italy.

(2) larger areas or entire estuarine areas. Moreover, erosion control measures should consider the environment energy level, distinguishing between low-energy areas and ocean-facing beaches, the latter being higher-energy systems. This distinction is useful for focusing on the best erosion control project, which is strongly site specific. Coastal erosion control methods can be mainly classified as structural and nonstructural measures. Structural measures can then be further classified as "hard" and "soft" alternatives. Structural measures involve permanent concrete, rock, or wooden engineering structures, since nonstructural measures generally refer to best management practices or to actions not involving man-made structures.

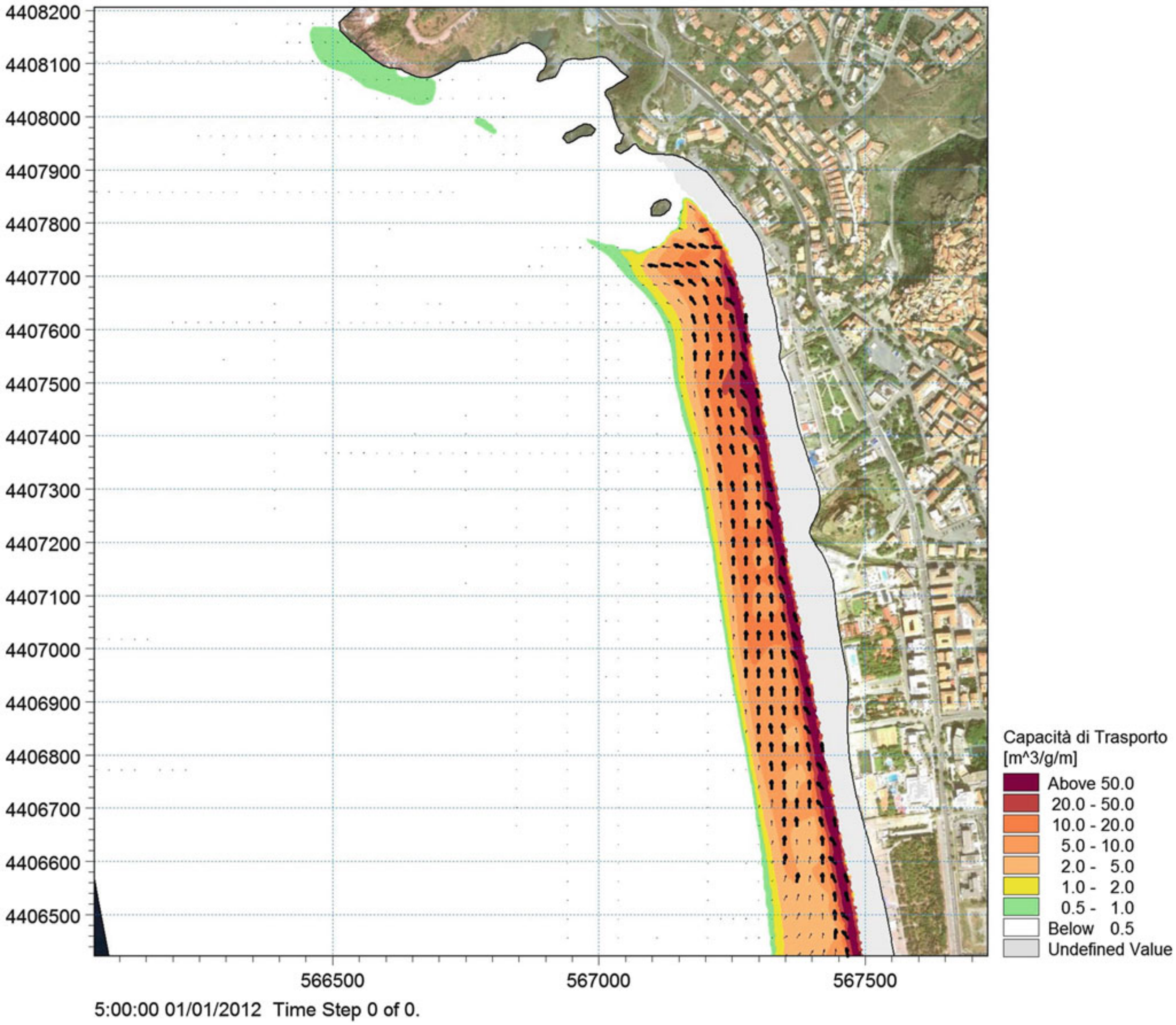
The main hard erosion control measures are:

- Breakwaters
- Revetments, seawalls, and bulkheads
- Groins

These structures are designed to protect the areas behind them by "fixing" the shoreline or by modifying the flow-field circulation by "trapping" sediment.

The main soft erosion control measures are:

- Nourishment
- Rip-rap, gabions, and paved-lining revetments
- Marsh sills
- Planting vegetation

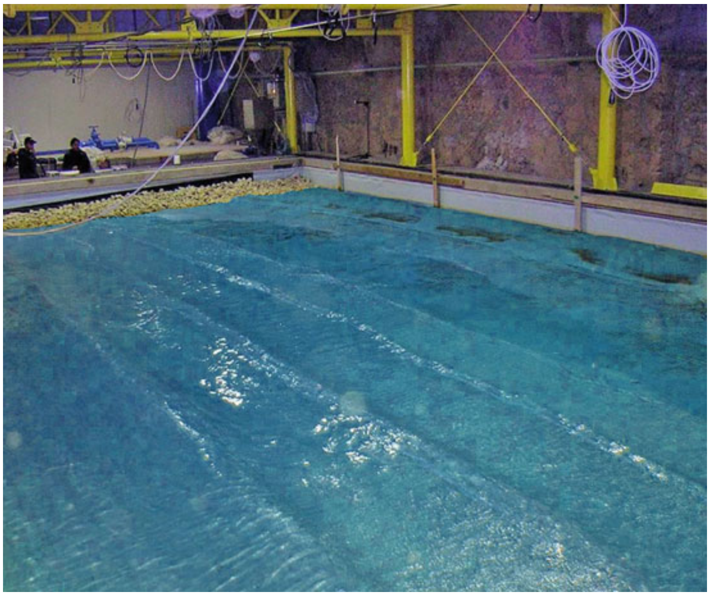


Coastal Erosion Control, Figure 2 Numerical results of bed load transport at Scalea beach, Italy.

Rip-rap, gabions, paved-lining revetments, groins, marsh sills, and planting vegetation are generally effective for river bank protection, while in wave-exposed areas, coastal erosion control measures should be chosen depending on whether cross-shore or long-shore transport is dominant, with a preference for shorel-parallel structures (such as revetments, attached and detached breakwaters) or perpendicular structures (such as groins), respectively. Furthermore, good coastal planning and management practices that promote land use rehabilitation targets and eco-sustainable tourism are also suitable coastal erosion control measures. Lastly, the choice of erosion control method is strongly dependant on the environmental, social, and cultural characteristics of the areas concerned and on legislative, policy, and economic aspects. Thus, the combination of different methods is often an effective strategy.

Structural measures: hard stabilization

Hard stabilization methods are based on protective structures designed to stabilize the shore and to prevent waves and tides from reaching the area or to trap sediment. Such structures typically act by reducing incident wave energy and changing flow-field circulation. Stabilization effectiveness is strongly dependant on wave exposure, the latter being influenced by structure inclination and orientation (e.g., perpendicular or parallel to the coastline),

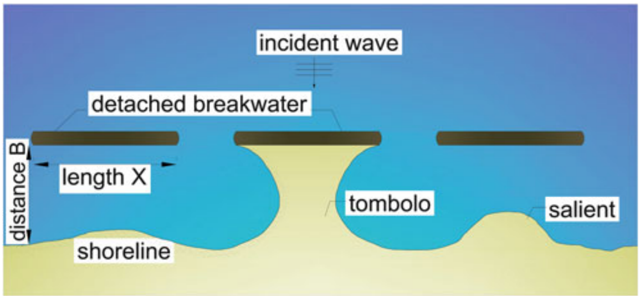


Coastal Erosion Control, Figure 3 Large basin simulation: example of an experimental investigation.

permeability/impermeability, overtopping, etc. The main hard structures are described below.

Breakwaters

Breakwaters (Figure 4) are structures designed to protect shorelines from erosion by shielding waves (i.e., reducing incident wave energy) and changing littoral transport conditions. These structures can be directly connected to the shoreline or constructed shore-parallel, respectively, attached or detached. In the first case, breakwaters act as a revetment, protecting adjacent upland areas against scour induced by waves and currents, while detached breakwaters act by allowing sand accumulation from the original shoreline to the landward breakwater. Such sand accumulation is called a “tombolo” or “salient,” depending on whether or not the breakwater is reached by sand. Accretionary beach features are characterized by a dimensionless ratio X/B (Herbich, 1991), X representing the breakwater length and B the breakwater distance from the original shoreline. For $X/B > 1$, sediment deposition and accumulation behind breakwater forms until the shoreline is connected to the structure (permanent tombolo), while for $X/B < 1$, sediment forms from the shoreline in the lee of the structure, without reaching the breakwater (salient). Since sediment transport phenomena are in equilibrium, sand accumulation leads to the formation of an erosion zone. With reference to design and construction aspects, breakwaters can be classified as follows: (1) rock or concrete units with trapezoidal cross sections, (2) prefabricated triangular-shaped concrete units, and (3) sand-filled containers (caissons) with geotextile units.

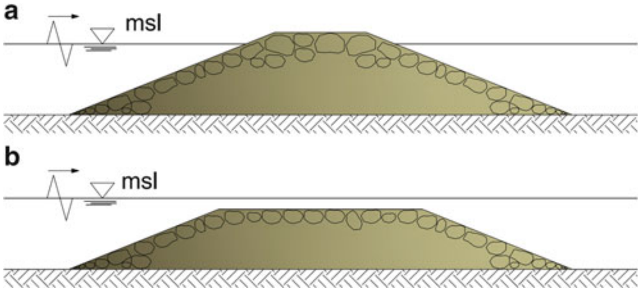


Coastal Erosion Control, Figure 4 Detached breakwaters: definition sketch.

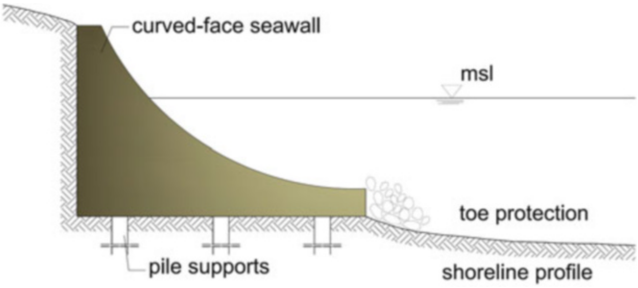
Breakwaters can be emergent and submerged, depending on crest position with respect to water level, namely, if the crest is positioned entirely below or above mean sea level (Figure 5). Both emergent and submerged breakwaters have a strong impact on sediment fluxes and on morphodynamic evolution, the latter depending on structure-induced waves, circulation fields, and wave overtopping. In order to avoid negative morphological effects, such as local scours, these aspects should be preliminarily and carefully analyzed. Lastly, it should be pointed out that during storm conditions, breakwater cannot stop or dissipate most of the waves, which result in low effectiveness, and thus other methods such as supplementary nourishment may be required.

Revetments, seawalls and bulkheads

Revetments, seawalls, and bulkheads are shore-parallel structures built adjacent to the shoreline. The main



Coastal Erosion Control, Figure 5 (**a**) Emergent; (**b**) submerged breakwaters.



Coastal Erosion Control, Figure 6 Seawall.

difference lies in their functional aspects (U.S. Army Corps of Engineers, 1984). In fact, revetments are stone or concrete structures built adjacent to the shoreline and are designed to protect the underlying soil from erosion. A larger stone layer is generally placed on the frontwater, with a smaller layer filter placed below it. The latter prevents underlying soil washing, while the main erosion protection is guaranteed by the upper stone layer.

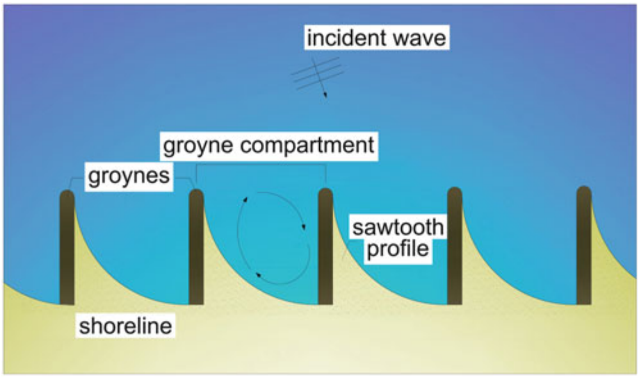
Seawalls are primarily designed to protect the shore against wave action; bulkheads are retaining walls designed to provide protection in low-to-moderate wave energy environments. More specifically, seawalls are massive concrete or stone, vertical or sloped structures (Figure 6), with rubble, curved, or stepped face. A combination stepped-curved face may also be constructed. Revetment and seawall slopes should be no steeper than 1:3, and the length of the structure should be carefully selected to avoid erosion of the adjacent coastline.

Bulkheads (Figure 7) are designed to protect adjacent upland areas and to retain the soil behind them. A sufficiently large embedded wall is required, and tie rods may be used to increase the stability of the structure. Toe protection is required for all these structures so as to prevent local scour.

Groins

Groins are long, narrow coastal structures used both on open beaches and in estuaries, altering nearshore tidal flow patterns and deflecting currents. They are placed perpendicular or slightly perpendicular to the shoreline (Figure 8). Such structures are generally constructed in groups, so that compartments between adjacent groins can be identified, trapping sediments in each of them and extending coast longevity.

Coastal Erosion Control, Figure 7 Wooden bulkheads (Rogers and Skrabal, 2001).



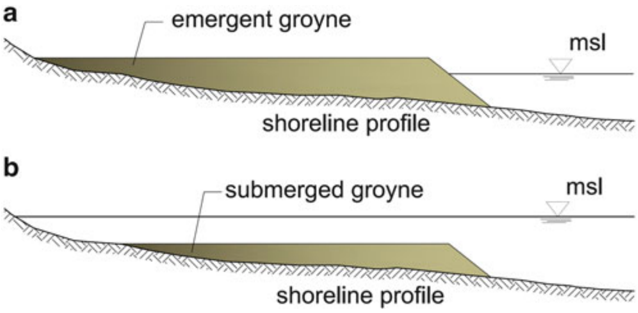
Coastal Erosion Control, Figure 8 Groins: definition sketch.

Groins are most effective when long-shore currents prevail in one direction.

Generally, two kinds of groins (Figure 9) can be distinguished (van Rijn, 2010; van Rijn, 2011):

- Impermeable, high-crested structures, usually made of sheet piling or concrete structures. Crest levels are 1 m above MSL (mean sea level). A full long-shore current block is expected, so that the sand within each compartment is retained. A typical sawtooth shoreline profile should be created, thus increasing groin spacing.
- Permeable, low-crested structures, with crest level between MLW (mean low water) and MHW (mean high water). These structures act as flow resistance, reducing the littoral drift in the inner surf zone. A regular shoreline should be created.

Groins should only be constructed along coasts with recession rates in excess of 2 m/year. Their length should moreover be extended over the inner surf zone



Coastal Erosion Control, Figure 9 (**a**) Emergent; (**b**) submerged groins.

(Basco and Pope, 2004; Kana et al., 2004). High-crested, impermeable groin length (*L*) and spacing (*S*) typically varies between 50 and 100 m and between 1.5 and 3 times the length of the groin, respectively (van Rijn, 2011). Finally, groins induce local scour at the toe of the structures and thus require regular maintenance.

Structural measures: soft stabilization

Nourishment

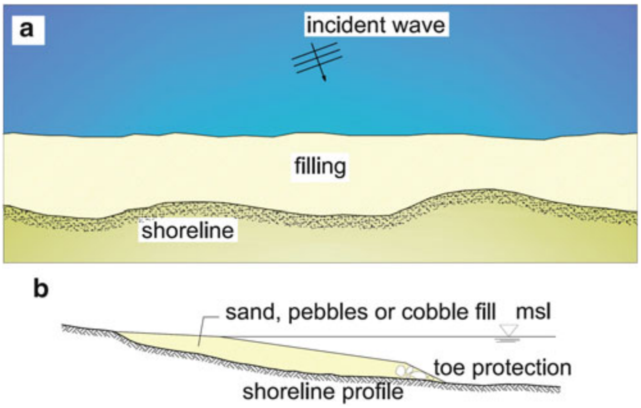
Nourishment is a shoreline stabilization method using sand, pebbles, or gravel beach fill (Figure 10). A key parameter of successful nourishment design is the choice of fill material. In fact, sediment should have the following main characteristics (Department of Boating and Waterways and State Coastal Conservancy, 2002):

- No contamination
- Fine grain size fraction
- Grain size comparable to or larger than in situ material

With reference to uncontaminated sediments, the introduction of contaminated material to coastal systems not only compromises habitats and ecosystems but also creates financial loss in terms of tourism, these areas being often used for recreational purposes.

With regard to the choice of sediment grain size, sediment with comparable or larger than in situ material size characteristics is preferable. In fact, comparably, grain size material tends to have the same behavior as in situ material, while larger sediment generally results in a more stable solution. Vice versa, fine grain size usually results in less stable solutions and accelerated erosion.

Sediment grain size and contamination level are strongly dependant on the source of the nourishment material. The latter generally includes dredged sediment from harbor construction and maintenance, lagoon restoration, and offshore and inland (e.g., damming rivers dredging) sources. Fill material may be placed (1) on the dry beach (dune nourishment); (2) on the beach cross section, on the dry portion and near the waterline, and across the entire beach cross section (i.e., above and below water); and (3) offshore as a sand bar (National Research Council, 1995).



Coastal Erosion Control, Figure 10 Nourishment: (**a**) plane view; (**b**) cross section.

In dune nourishment configuration 1, fill material is placed high above the waterline. This configuration provides effective protection against storm waves, but no expansion in dry beach width and no increase in recreational coastal areas. In configuration 2, an immediate increase in beach width (i.e., recreational areas) is observed. Furthermore, once placed, fill material is redistributed offshore and alongshore below wave and current action until a stable configuration is achieved. If fill material is placed both above and below the waterline, an already stable configuration is attempted, and there is little offshore sand redistribution, which leads to minimal changes in dry beach width. Finally, in configuration 3, fill material is placed in the surf zone, and the sand gradually moves onshore below wave and current action, thus increasing the beach width.

In regard to environmental aspects, nourishment can have a strong impact on aquatic habitats on the seabed, near the shoreline. Thus, species ecosystem response tolerance and the burial adaptability thereof should also be considered. In all cases described above, because of the nourishment design characteristics, waves and currents gradually remove some of the sediment, and periodic maintenance is required.

Rip-rap, gabions, and paved-lining revetments

Shoreline revetments may be constructed using rip-rap revetments (Figure 11), gabions, and paved linings (Figure 12) that are wire cages filled with stones and placed as revetment along the shoreline and river banks, in vertical stacked or sloped configurations.

Marsh sills

Marsh sills are shore-parallel structures designed to protect planted wetland vegetation. An offshore wood or rock mound (sill) and an intertidal area are created between the sill and upland (Figure 13). Protection is achieved thanks to existing or planted vegetation in the intertidal zone, which dissipates wave energy, preventing it from reaching the upland (Rogers and Skrabal, 2001). An added value of this strategy is its ability to promote the creation of natural habitats.

Planting vegetation

Vegetation plays an important role in promoting shoreline stabilization, reducing wave and current energy, and trapping incoherent sediment in radical apparatuses. Thus, planting vegetation is considered a shoreline stabilization method, although its effectiveness is strongly site specific. In fact, planting vegetation generally allows good erosion control to be achieved in low-energy environments, such as in estuarine tidal zones, while on the contrary, in high-energy environment, vegetation seems ineffective. Finally, there should be a preference for the planting of native species.

Coastal Erosion Control, Figure 11 Rip-rap revetments (Trowell, 2012).

Nonstructural measures

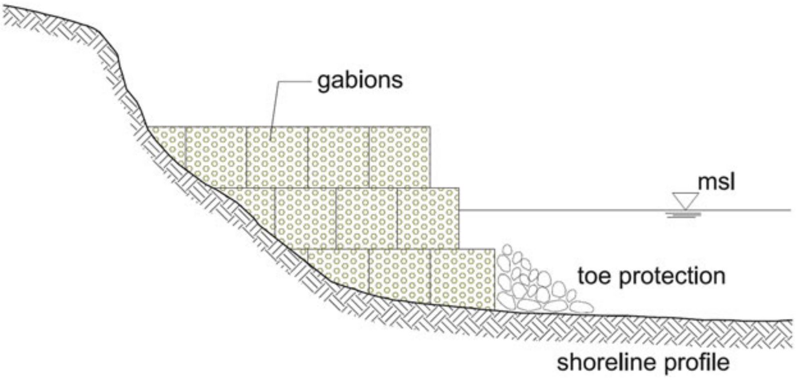
Policy and planning techniques

Policy and planning techniques for erosion control relate to strategies for coastal area use and anthropogenic pressure control, based on the introduction of sustainable coastal area management development logic. In fact, man's use of coastal and estuarine areas for promoting economic activities generally requires intensive development and accelerated estuarine area modification processes, often leading to natural equilibrium alteration and increased vulnerability. Policy planning techniques involve a large number of factors, the majority of which are described herein.

Firstly, estuarine area management project measures are based on specific environmental policies, legislations (e.g., European Bird, 1979 and Habitats, 1992 Directives, Natura 2000 ecological networks of protected areas to name but a few), and project management development. The latter is generally based on "prevention" and "protection" measures, attending to primary coastal erosion risk management and consisting of both structural and nonstructural measures (Safecoast, 2008). Examples of "prevention" strategies are relocation, zoning, space allocation and reservation, coastal erosion risk education, and communication and raising awareness. Examples of "protection" strategies are based on building and maintaining structures for erosion control (such as breakwaters, nourishment and groins).

Good project management should be based on a precise site evaluation and decision logic systems, with focus on the following aspects (Safecoast, 2008):

- Physical and environmental characteristics
- Economic and ecological values, assessed by stakeholders and others, such as engineers, scientists, politicians, land use planners, and the public affected
- Historical and cultural background
- Policy measures and a general set of rules capable of identifying different scenarios (e.g., land use restriction)



Coastal Erosion Control, Figure 12 Gabions: definition sketch.

Coastal Erosion Control, Figure 13 Marsh sills (Modified from Trowell, 2012).

- Coastal dynamics, effectiveness of erosion control measures, erosion phenomena, and spatial and temporal scales

On this basis, the resulting project will be the “optimum solution,” chosen in continuity with existing management policies, protection measures, and related operational procedures, such as institutional arrangement, operational responsibilities, and financing.

Conclusions

Natural and anthropogenic factors cause alterations in coastal sediment transport equilibrium, thus resulting in coastal erosion, with ecological and financial loss. Furthermore, cultural identity, resources, and recreational coastal land use are strongly compromised or lost.

Hard and soft control measures are effective tools for mitigating such phenomena. Hard stabilization methods in particular mainly consist of concrete and rock structures such as groins, revetments, breakwaters, etc., while soft methods consist of shoreline protection measures based on beach nourishment and the planting of vegetation, thus allowing recreational tourist areas to be developed. Good practices in the planning and management of coastal areas are also effective coastal erosion control measures and important targets for many states. The choice of stabilization methods should be the “optimal” method among possible solutions and should be selected starting from a hydrodynamic, environmental, social, and cultural site characterization. Finally, the combined use of different stabilization methods enables considerable coastal erosion control objectives to be achieved.

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Cross-references

Bulkheads
Revetments
Sediment Grain Size
Shore Protection

COASTAL INDICATORS

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Synonyms

Coastal state indicators

Definition

Quantitative/qualitative statements or measured/observed parameters that can be used to describe existing situations and measure changes or trends over time (UNESCO, 2006, p. 11), concerning the state of an estuary or other coastal feature.

Aims and concepts

Indicators aim to convey a complex message in a simple manner. Their three main functions are to simplify the information, quantify the target system, and provide a facilitator tool in the communication process between different stakeholders (UNESCO, 2006). For this reason, coastal indicators constitute an extremely useful tool in coastal management since they can translate observations, models, and scientific interpretation – which are too complex and difficult to be used directly in the managing processes – in a simplified form to coastal managers. Therefore, indicators facilitate the integration of scientific know-how in coastal zone planning and management (e.g., UNESCO, 2003; UNESCO, 2006; NOAA, 2010) and reduce the risk of failure in the communication process between scientists and coastal managers (e.g., van Koningsveld, 2003; van Koningsveld et al., 2005; Jiménez, 2010). Thus, in the determination of adequate coastal indicators to describe a particular system, it is fundamental to receive the input of coastal managers and scientists in a joint effort to define the adequate indicators (van Koningsveld, 2003). The former are able to assess what information will be of most value to the management, while the latter can determine what might be possible to measure based on existing or potential technology and scientific understanding (van Rijn, 2010). As an example, shoreline position is one of the most commonly used indicators to determine coastal morphodynamic state.

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COASTAL LAGOONS

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Synonyms

Coastal bays; Coastal lakes; Coastal ponds

Definition

Coastal lagoons are shallow brackish or marine water bodies separated from the ocean by a barrier island, spit, reef, or sand bank (Colombo, 1977; Barnes, 1980; Kjerfve, 1994; Kennish and Paerl, 2010a). Depending on the extent of the barriers, they may be partially or totally enclosed, although most are connected at least intermittently to the open ocean by one or more restricted tidal inlets. Oertel (2005) called the smaller, totally enclosed systems coastal lakes or coastal ponds. Those with outlets to the sea are termed coastal lagoons and coastal bays, depending on their shapes.

Introduction

Coastal lagoons form on low-lying coasts such as along the Atlantic and Gulf coasts of the USA, where they are particularly extensive, covering ~2,800 km of shoreline (Nichols and Boon, 1994). They are much less common on most other coasts, occupying only ~12 % of the coastal shorelines worldwide. The Antarctic is the only continent devoid of coastal lagoons, while they are most prominent along the coasts of Africa (17.9 % of the coastline) and North America (17.6 %) and less conspicuous along the coasts of Asia (13.8 %), South America (12.2 %), Australia (11.4 %), and Europe (5.3 %) (Barnes, 1980; Kennish and Paerl, 2010a).

The size and shape of coastal lagoons vary considerably, although they are usually oriented with their long axis parallel to the shoreline, as exemplified by the Barnegat Bay-Little Egg Harbor system in New Jersey (USA) (Figure 1) (Kennish, 2001). However, some lagoonal water bodies have a triangular or delta shape with v-shaped landward margins, as demonstrated by the Rehoboth Bay and Assawoman Bay in Delaware (USA) (Oertel, 2005). They range in size from a few square kilometers up to 10,000 km^2 as in the case of the expansive Lagoa dos Patos in Brazil (Bird, 1994).

Formation

The genesis of coastal lagoons is closely linked to the formation of coastal barriers which separate flooded basins landward from the coastal ocean. According to de Beaumont (1845), the barriers form by the upbuilding of bars and shoals. Gilbert (1885) attributed barrier formation to the progradation of spits which creates shallow embayments behind them. McGee (1890) advanced an inundation model of coastal lagoon formation whereby a rising sea floods lowland areas. Oertel (2005) supported the models of Gilbert (1885) and McGee (1890) as the two main modes by which coastal lagoons form.

Physical-chemical characteristics

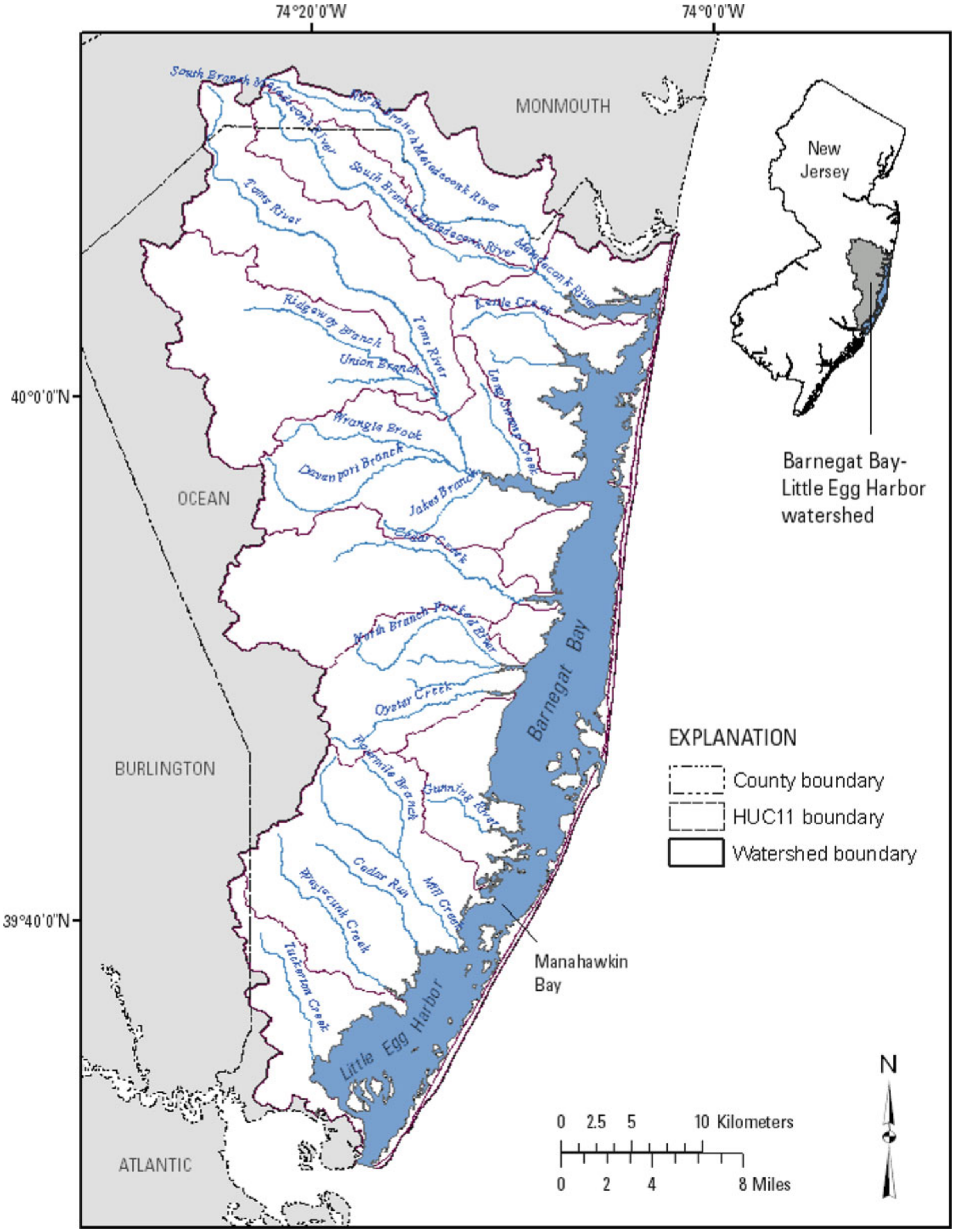
The basin morphometry and circulation of coastal lagoons differ considerably from those of larger, river-dominated estuaries. Coastal lagoons are shallow, generally averaging less than 2–3 m in depth, but depths of up to 30 m have been recorded in some tidal channels of these systems (Oertel, 2005; Kennish and Paerl, 2010b). They are generally well mixed by wave and current action. Because coastal lagoons receive relatively small volumes of freshwater input, tidal exchanges through narrow inlets play a significant role as a driver of lagoonal circulation. Most coastal lagoons are microtidal systems.

The physical-chemical processes taking place in coastal lagoons depend greatly on multiple factors, notably the size and configuration of the tidal inlets, expanse and development of bordering watersheds, amount of freshwater input, tidal prism, wind velocity and direction, and water depth (Alongi, 1998; Kennish and Paerl, 2010a). As stated by Kennish and Paerl (2010a), "Variations in precipitation and evaporation, surface runoff, and groundwater seepage, together with fluxes in wind forcing, account for large differences in advective transport in lagoonal estuaries. Storm and wind surges, overwash events, inlet configurations, land reclamation, construction of dams, dikes and artificial bars, as well as channel dredging events, are important drivers of hydrological change in these systems."

Because of the extreme enclosure of most coastal lagoons by barriers and the limited tidal exchange with ocean waters, these shallow systems tend to have protracted water residence times. As a result, coastal lagoons are susceptible to accumulation of pollutants from coastal watersheds and airsheds. They are also easily impacted by overwash events driven by extreme climate events such as hurricanes that can transport large amounts of beach and coastal ocean sediments into these backbays. This was the case in New Jersey when superstorm Sandy made landfall on October 29, 2012, creating a storm surge exceeding 4 m in some areas and dumping more than 1.5 million cubic meters of beach sand into Barnegat Bay-Little Egg Harbor. Similar events have been recorded for other coastal lagoons impacted by hurricanes and extratropical storms.

Sediments

Coastal lagoons receive terrigenous sediment from streams and rivers draining coastal watersheds. These sediments often consist of fine silts and clays, much of which flocculate and are deposited at the mouth of the influent systems. Fine-grained sediments also accumulate near the lagoonal shoreline in proximity to salt marshes which facilitate deposition of silts and clays. However, in some



Coastal Lagoons, Figure 1 Barnegat Bay-Little Egg Harbor, a coastal lagoon located along the central New Jersey coastline (USA). Note the coastal watershed draining to the lagoon and the barrier island system forming the eastern boundary. Figure 1 from US Geological Survey, West Trenton, New Jersey.

coastal lagoons, the influx of sediments from land sources is minimal, and a significant amount of sediment accumulating in various areas of the lagoonal basin is the result of sediment reworking of the lagoonal floor. This is the case in Barnegat Bay-Little Egg Harbor, New Jersey (Psuty, 2004; Psuty and Silveira, 2009), as well as many other temperate coastal lagoons of North America (Oertel, 2005). Coarser sediments generally are found in proximity to the backbarriers and tidal inlets. These sediments, which are typically better sorted than those near the mainland, primarily derive from marine and backbarrier sources via storm surge and overwash events which build washover fans, and tidal currents through inlets which build ebb-tidal deltas and other sandy deposits in the lagoonal basin.

Biotic production

Coastal lagoons are characterized by high levels of biotic production. This is so because the photic zone extends to the lagoonal floor in most areas, and they usually receive considerable amounts of nutrients from the surrounding watersheds which stimulate primary production. Benthic algal and seagrass production can exceed phytoplankton production in coastal lagoons. In addition, there is strong benthic-pelagic coupling; in coastal lagoons the effects of biogeochemical cycling, bioturbation, and other interactions between the bottom sediments and the overlying water column may be far greater than those in deeper estuaries. Nutrients may be recycled many times before exiting inlets to the coastal ocean due to protracted water residence times which account for high rates of productivity per unit nutrient input (Kennish and Paerl, 2010b).

The range of annual primary production in coastal lagoons is large (~50–>500 g C m^{-2} $year^{-1}$). Based on the classification of Nixon (1995), many coastal lagoons fall within the range of eutrophic conditions (300–500 g C m^{-2} $year^{-1}$) or even exhibit hypereutrophic conditions (>500 g C m^{-2} $year^{-1}$) (Nixon, 1995). The high primary production in these water bodies, together with the input of organic matter from adjoining wetlands and external systems, supports rich faunal communities, with many species utilizing these environments seasonally. Benthic macrofaunal productivity in coastal lagoons amounts to ~20–200 g ash-free dry weight m^{-2} $year^{-1}$, with zooplankton productivity being as much as 50 % of this amount. Nekton productivity in turn ranges from ~10 % to 100 % of the zooplankton productivity in these systems (Alvarez-Borrego, 1994). Coastal lagoons also provide ideal nursery and feeding habitats for many marine fauna (Kennish and Paerl, 2010b; Day et al., 2012).

Anthropogenic effects

Coastal lagoons are used for fisheries and aquaculture, energy production, biotechnology, transportation, shipping, and many other human uses (Pauly and Yáñez-Arancibia, 1994; Kennish and Paerl, 2010b). Watersheds surrounding coastal lagoons are often heavily populated and developed because of the great commercial and recreational value of these water bodies, their exceptional ecosystem services, and the access they afford to coastal ocean waters. However, altered land use/land cover of upland areas associated with increasing population growth and development, together with escalating human activities in the coastal lagoons themselves, has impacted their structure and function and compromised their ecological integrity (Kennish and Paerl, 2010b). For example, the removal of natural vegetation, compaction of soils, and construction of impervious surfaces promote nutrient runoff into the lagoons, hastening their nutrient enrichment and eutrophication (Kennish, 1997; Kennish, 2002).

Eutrophication of coastal lagoons and estuaries is on the increase worldwide (Nixon, 1995; Kennish et al., 2008; Kennish, 2009; Kennish and Paerl, 2010a), and it poses the greatest threat to the ecological integrity of these valuable ecosystems (Kennish and de Jonge, 2011). Eutrophication leads to an array of cascading changes in ecosystem structure and function such as decreased dissolved oxygen levels, increased microalgal and macroalgal abundance, occurrence of harmful algal blooms (HABs), loss of seagrass habitat, reduced biodiversity, declining fisheries, imbalanced food webs, altered biogeochemical cycling, and diminished ecosystem services (Nixon, 1995; Kennish, 1997; Kennish et al., 2008; Kennish and Paerl, 2010b).

Because of their extreme enclosure and restricted circulation, coastal lagoons are highly susceptible to accumulation of chemical contaminants such as polycyclic aromatic hydrocarbons, halogenated hydrocarbons, and metals. Bottom sediments serve as a repository and secondary pool of these hazardous substances. Volatile organics and plastics are also a potential threat to organisms inhabiting these environments. Oil spills are particularly detrimental. Pathogens delivered to lagoonal systems in stormwater runoff subsequent to rainfall events frequently compromise their water quality, although such events are usually ephemeral.

The shorelines of many coastal lagoons are altered by housing and bulkhead construction, which interferes with natural processes and directly impacts habitat. The siting of marinas along these shorelines, oil and gasoline leakages from fixed installations, sanitation-tank releases from boats, sewage wastewater discharges, and dredging activities adversely affect lagoonal organisms. Aquaculture operations can markedly degrade water quality in confined areas. In many systems, organic loading contributes to elevated BOD levels and significant oxygen depletion leading to system impairment.

Conclusions

Coastal lagoons are highly productive, enclosed water bodies that are heavily utilized by humans. They are complex physiographic features susceptible to eutrophication and other anthropogenic impacts due to their relatively low freshwater inputs, shallow depths, restricted circulation, poor flushing, limited ocean exchange, and protracted water residence times. As a result, coastal lagoons are beset by similar problems such as depleted dissolved oxygen, habitat loss and alteration, and, in some cases, altered ecosystem structure and function. Indicators of eutrophication are widespread in these shallow water bodies, including elevated chlorophyll *a* levels, HABs, submerged aquatic vegetation loss, and impacted biotic communities and harvestable fisheries. Progressive eutrophication of coastal lagoons can lead to permanent loss of essential habitat, diminished aquatic life support, and a marked decline in human use. Because of their enclosure, coastal lagoons are also susceptible to chemical contaminant inputs, pathogens, and organic carbon loading. The hardening of lagoonal shorelines, constructing of installations, and dredging of sediments physically alter habitats which also impacts biotic communities and their sustainability.

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Cross-references

Backbarrier
Barrier Island

COASTAL LANDFORMS

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Synonyms

Coastal geomorphic forms

Definition

Coastal landforms are those formed and modified by various geological and oceanographic processes. The present-day coastal land was carved out during the Late Quaternary, particularly during the Holocene. Anthropogenic activities enhance the coastal landform changes.

Introduction

The coasts are dynamic and their morphology is continually changing in response to various processes operating at different rates. Climate change and sea-level variations during the Quaternary period have strongly influenced the geomorphic and sedimentation processes in the coastal regions, and much of the coastal land was carved out during the Late Quaternary period. Sea-level changes during the Holocene have influenced the evolution of coastal environments such as estuaries/lagoons and barrier complexes and controlled the sedimentation in the coastal environments (Narayana and Priju, 2004). The evolution and subsequent changes of coastal landforms were influenced by various factors, viz., the coastal processes, sea-level changes, and tectonics. These landforms are modified by a variety of dynamic processes and the driving forces include framework geology, oceanographic processes, river-mouth processes, sediment supply, and human activity (FitGerald et al., 2008). Coastal landforms are extremely variable and coastal habitats change over a range of spatial and temporal scales, and recognition of these variations is necessary for effective planning and management (Woodroffe, 2007). Barrier islands, wetlands, and other parts of coastal systems might have a threshold, and, when the limits of threshold are exceeded, the landforms become unstable and prone to irreversible changes in form and position (Williams and Gutierrez, 2009).

Coastal systems exhibit two distinct types of coastal landforms: depositional and erosional. Erosional coastal landforms typically exhibit high relief and rugged topography, which include sea cliffs, wave-cut platforms, and stacks. The depositional coastal landforms include barrier islands, beach ridges, cheniers, tidal flats, mudflats, etc. In this chapter, we focus on the depositional coastal landforms and their characteristics.

Barrier Islands

Coastal barriers and spits are often regarded as similar coastal forms in terms of beach deposition projecting across coastal bays. While barriers tend to bridge the bay by joining the mainland at each end, spits are only attached at one end. However, many barriers show cross-barrier breaks or breaches through which the sea may enter on a permanent or intermittent basis, thus forming barrier islands (Figure 1). Coastal barriers are complex constructional morphological features involving deposition by waves, wave-generated currents, tidal currents, and wind activity (Hayes, 1979). A barrier exhibits two morphodynamic units – a seaward beach face and a landward facing back-barrier slope – and these two units develop when the barrier is gravel dominated (Orford et al., 1996). As sand becomes the dominant component, a third environment comprising aeolian dunes can appear at the top of the beach face (barrier crest) and spread onto the backslope. Current flows may have been responsible for the initial submarine platform under the barrier, but with wave action forcing, onshore migration of the barrier takes place in combination with fine sedimentation characteristics of the low-energy back-barrier bay. Tidal currents also become dominant once barrier islands appear.

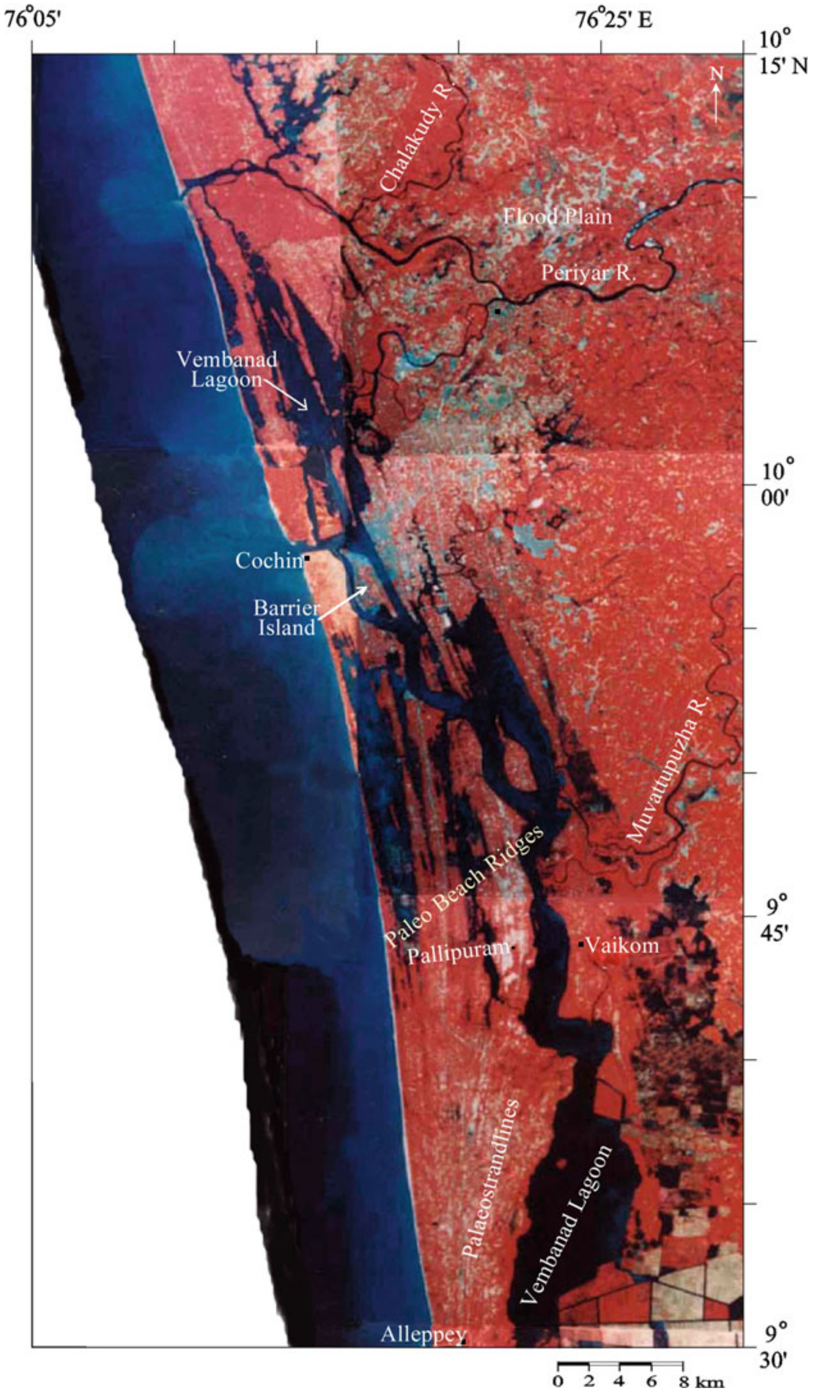
Sediment supply and the type of sediment are major controls on barrier development with which a behavioral distinction can be drawn between sand-dominated barriers and gravel-dominated barriers. This distinction has a spatial basis with gravel-dominated barriers being more prevalent in mid-upper latitudes compared to sand-dominated barriers, which reflects the greater potential of coarse material in high latitudes as a residue of late Quaternary glacigenic processes.

Coastal bars

Coastal bars can be broadly defined as aggradational ridges of sediments whose formation, morphology, and behavior are determined by interactions between waves, currents, tides, local slope, and grain size. Bars occur along beach, river delta, estuary, and continental shelf environments with a wide range of sizes, types, and orientation (Figures 1 and 2). Beach morphology undergoes cycling change, promoting offshore sediment transport and bar formation during winter; while during summer when the oceanographic conditions are calmer, the landward migration of the bar and eventual welding to the beach face take place. However, the existence of such "winter" and "summer" profiles is not universal, as both barred and non-barred profiles occur at all times in some areas, while in others only one type may persist throughout the year. Furthermore, cyclic beach response at timescales much shorter than seasons can result in barred profiles (Short, 1979). Types of bars are often distinguished based on their alongshore planform shape and orientation relative to the shoreline as linear, shore-parallel, sinuous, or crescentic with a trough separating them from the shoreline. Some of the coastal areas consist of alternating transverse bars, welded to the shoreline and are separated by channels occupied by rip currents (Figure 2). Bar type is strongly related to wave energy level with linear bars developing under high-energy conditions, crescentic bars during intermediate energy, and transverse bars during lower wave energy levels. Under very low-energy conditions, a bar may become fully welded to the beach and appear as a flat terrace at low tide. These types of bar configurations are common on microtidal beaches and may grade into each other as energy levels vary. A number of classifications exist describing both bar types and the continuum of bar evolution (e.g., Greenwood and Davidson-Arnott, 1979; Short and Aagaard, 1993; Wijnberg and Kroon, 2002).

Coastal lagoons

The term lagoon describes a stretch of salt water separated from the sea by a low sandbank or coral reef (Figures 1 and 3). Coastal lagoons (Figure 3) are mostly estuarine, usually shallow, and have generally been partly or wholly sealed off from the sea by the deposition of spits or barriers, by localized tectonic subsidence, or by the growth of coral reefs. They are best formed on transgressive coasts, particularly where the continental margin has a low gradient, and sea-level rise is slow. The lagoons are ephemeral features and their depths and areal extent gradually decrease due to sedimentation from inflowing rivers, as well as accumulation of sediment washed in from the sea, wind-blown material, and chemical and organic deposits. Lagoons range in size from less than a kilometer to more than a 1000 km, and they occur on about 12 % of the length of the world's coastline (Bird, 2000). They can be classified on the basis of infilling or increasing in size (Nichols, 1989). The infill of some



Coastal Landforms, Figure 1 Satellite image showing various coastal landforms such as barrier island, lagoon, paleo-strandlines, and flood plains along southwest coast of India.

lagoons, particularly those that are parallel to the shore, may involve the development of cuspate forms that divide the lagoon into a series of segments. These divisions have been attributed to winds blowing along the length of the lagoon producing waves which build spits that isolate the lagoon into separate basins.

Mudflats

Mudflats (Figure 4) occur along low-energy shorelines that are well supplied with silt and clay-sized sediments, particularly on many estuarine margins, delta shorelines, and areas of open coast subject to low wave energy. Such settings are usually dominated by tidal processes, and the characteristic

Coastal Landforms, Figure 2 Well-developed shore-attached transverse bars and adjacent deeper rip channels at Lighthouse Beach, New South Wales (Short, 1979) (Source: http://www.ozcoasts.gov.au/conceptual_mods/beaches/wdb.jsp).

Coastal Landforms, Figure 3 Hypersaline coastal lagoon at Mar Menor, Iberian Peninsula, Spain (Source: http://www.latorreholiday.co.uk/3.html).

landforms of muddy coasts – salt marshes, mangrove swamps, and tidal flats – are often well developed under macrotidal conditions (Hayes, 1975). Enormous quantities of muddy sediment are supplied by some of the world's major rivers, and their estuaries and deltas often feature extensive shore-attached mud banks. Consequent to the development of wide estuarine mouths, a lot of seawater enters through them during high tide and submerges low-lying flats adjacent to the river mouth forming tidal flats. Both estuarine and open coast mud banks are highly dynamic landforms, which exhibit seasonal variability in response to variations in river flow and wave energy.

Fine sediments in suspension can be transported over long distances by coastal currents. These fine particles undergo flocculation and, once they are flocculated, settle from suspension rapidly giving rise to muddy deposits/mudflats near estuarine/river mouths. Flocculation is influenced by a variety of factors, notably salinity, fluid shear, and suspended sediment concentration (Lick and Huang, 1993). The effect of these processes may vary over quite short spatial and temporal scales, especially in estuaries, where mixing of freshwater and saltwater occurs and where marked variation in flow intensity occurs at tidal timescales. The cohesive nature of muddy sediments makes

Coastal Landforms, Figure 4 Present-day mudflats of Sado River Estuary, Portugal (Source: http://geologicalintroduction.baffl.co.uk/?p=323).

their behavior far more complex than that of non-cohesive sands. Flocculated sediments typically settle from suspension far more rapidly than their constituent mineral particles, and the stability of natural muddy deposits is governed not only by physical processes but also by the activity of a rich and diverse biota (Paterson, 1997).

Beaches

Beach is a wave-dominated accumulation of sediment located between wave base and the upper swash limit. A beach system is a product of the interaction of waves, tide, and sediment, and hence, beaches exist in a wide spectrum of wave, tide, and sediment combinations and geological settings. Beach systems occur in all tide ranges, in all latitudes, and in all climates. Beaches composed of fine sand through boulders may range from low-energy to high-energy systems exposed to persistent 2–3-m-high swell which breaks across wide surf zones. All beaches contain three dynamic zones – wave shoaling, wave breaking, and swash–backwash. The wave shoaling zone extends from the modal wave base where average waves can entrain and move sediment shoreward, to the outer breakpoint. The wave shoaling zone is dominated by asymmetrical wave orbital motions which produce a concave upward profile. It extends out to depths of 30 m or more which may lie 2–3 km offshore on high-energy coasts, while on low-energy coasts, it may only extend to low tide – a few meters from the shore. The surf zone, located between the breakpoint and shoreline, has the greatest potential for complex dynamic processes and resulting topography and bedforms. The width of the surf zone depends on the surf zone gradient, a function of sand size and wave height. The width may vary from a few meters on a steep reflective beach, typically 50–100-m-wide on a single bar intermediate beach, and up to several hundred meters on a high-energy dissipative beach. Surf zone topographic features include shore-parallel bars and troughs with waves breaking over the bars and reforming in the troughs. Surf zone bedforms reflect the changing velocity and direction of currents and depth of water and range from flat bed over the shallow bars to wave orbital and shore perpendicular current ripples in the troughs, to shore-parallel seaward migrating ripples in the rip channels (Short, 1979).

Beaches are of three types, which refer to the morphodynamic character of a beach system: wave-dominated, tide-modified, and tide-dominated beaches. Wave-dominated beaches occur where waves are high relative to the tide range. This can be defined quantitatively by the relative tide range (Masselink and Short, 1993)

$$\mathrm{RTR} = \mathrm{TR}/\mathrm{Hb} \qquad (1)$$

where TR is the spring tide range and Hb the average breaker wave height. When $\mathrm{RTR} < 3$, beaches are tide dominated; when $3 < \mathrm{RTR} < 15$, they are tide modified; and, when the $\mathrm{RTR} > 15$, they become tide dominated.

Shingle beach

The term “shingle” has been used to describe sediments composed of mainly rounded pebbles, larger in diameter than sand (>2 mm) but smaller than boulders (<200 mm) (Figure 5). In many locations, shingle is mixed with sand, silt, clay, or organic debris, resulting in a “mixed” sediment beach (Kirk 1980), but all shingle and boulder beaches can be regarded as different types of “coarse clastic” beach (Carter and Orford, 1991). Shingle coasts form in wave-dominated locations where suitably sized material is available, and they occur in high latitudes and temperate shores, which were affected by Quaternary glaciation.

Coastal Landforms, Figure 5 Sand and shingle beach, Blakeney Point, Sheringham (Source: http://www.geograph.org.uk/photo/2019176).

Coastal Landforms, Figure 6 Low amplitude beach cusps on Panambur beach, near Mangalore, west coast of India. Wide surf zone is seen on the background (Picture by K.S. Jayappa).

In general, shingle coasts have received less scientific attention than sandy and muddy shorelines, as they are much less common worldwide. However, recently there has been an increasing awareness of the geomorphologic, ecological, and engineering significance of shingle coasts in the contexts of sea-level change, flood defense, and habitat conservation (Packham et al., 2001).

Beach cusps

Beach cusps are crescentic, concave-seaward, and regularly spaced features occurring along the shorelines (Figure 6). The term "beach cusp" has been used for features with spacing ranging from 10 cm to many hundreds of meters; the term "swash cusp" has been used for rhythmic beach features with a spacing less than tens of meters (Hughes and Turner, 1999). Beach cusps are most commonly associated with medium to coarse sands, shingle, or mixed sand–shingle sediments on steep beaches, demonstrating significant wave reflection (Nolan et al., 1999). Multiple sets of cusps may be present at different levels on beaches of high tidal range. Beach cusps consist of embayments or swales separated by triangular horns which are normally comprised of coarser sediments. Under low-energy conditions, oscillatory flows, horn

Coastal Landforms, Figure 7 *Beach ridges* indicating the past sea level and abundant sediment supply along southwest coast of India.

divergent flows, and horn convergent flows will develop, and under high-energy conditions, sweeping flows and swash-jet flows can occur (Masselink et al., 1998).

Beach ridges

Beach ridges are azonal accumulation forms developed on seashores. They are usually subparallel ridges of sand, gravel, or pebble, as well as detritus of shell, situated in the foreshore zone, which is the boundary of low and high water range (Figure 7). Older complexes of beach ridges appear in the backshore zone, which lies above the high water range. Beach ridges forming at the present day are roughly parallel to the coast. Two types of beach ridges may develop on a progradational sea coast (Carter, 1986). The first type is a result of gradual accretion and coalescing of swash bars during transport of a deposit by wave action, and the second type is connected with longshore bar emergence during low wave energy conditions and simultaneous fall of sea level. They are constructed mainly by landward dipping laminae, and the morphology of these ridges is more complicated.

Beach ridges are also partly developed by the processes of aeolian deflation and accumulation. There is often an accumulated cover of aeolian deposits on earlier formed ridges, stabilized by vegetation. As a result, on the beach ridges, irregular hummock dunes or parallel foredune ridges can be situated (Carter and Wilson, 1990). Beach ridges are good paleogeography indicators of past wave regimes, sediment supply, sediment source, climatic conditions, sea-level change, and also isostatic emergence or submergence of land. Hence, beach ridges can be used to reconstruct past relative sea-level changes and the history of deposits.

Chenier ridges/plains

The name “chenier” derives from the French word chene, meaning oak, which grows on the coast of Louisiana, USA. Chenier ridges (cheniers) are elongated beach ridges with sand or shell composition and are separated laterally from other cheniers on a chenier plain, by fine grained sediments (Figure 8). Chenier ridges frequently bend landward at the downdrift end and branch in a fan-like fashion. Cheniers are found on generally low wave energy, low gradient, muddy shorelines, and in areas where there is an abundant sediment supply (Augustinus, 1989). Cheniers can be up to 6 m high, tens of kilometers in length, and hundreds of meters wide.

Raised beaches

A raised beach is a relict depositional landform comprising mostly wave-transported sedimentary material and preserved above and landward of the active shoreline (Figure 9). Raised beaches were first described by Jamieson in 1908, when he stated that raised beaches form along coasts or lake shorelines and are well recognized as indicators of a fall in relative sea level. In certain situations, multiple raised beaches may form adjacent to one another, producing a beach ridge plain or strandplain (Otvos, 2000). The elevated position of a raised beach relative to active shoreline processes may be the product of one or more of the following mechanisms: (1) tectonic uplift (Garrick, 1979), (2) isostatic rebound related to ice-unloading of a land mass (Smith et al., 2000), (3) depositional regression involving delivery of sediment to a shoreline at a rate sufficient to allow formation and stranding of successive beaches (Thom, 1984), and (4) forced regression whereby eustatic sea-level fall leads to abandonment of a shoreline (Murray-Wallace and Belperio, 1991).

Spits

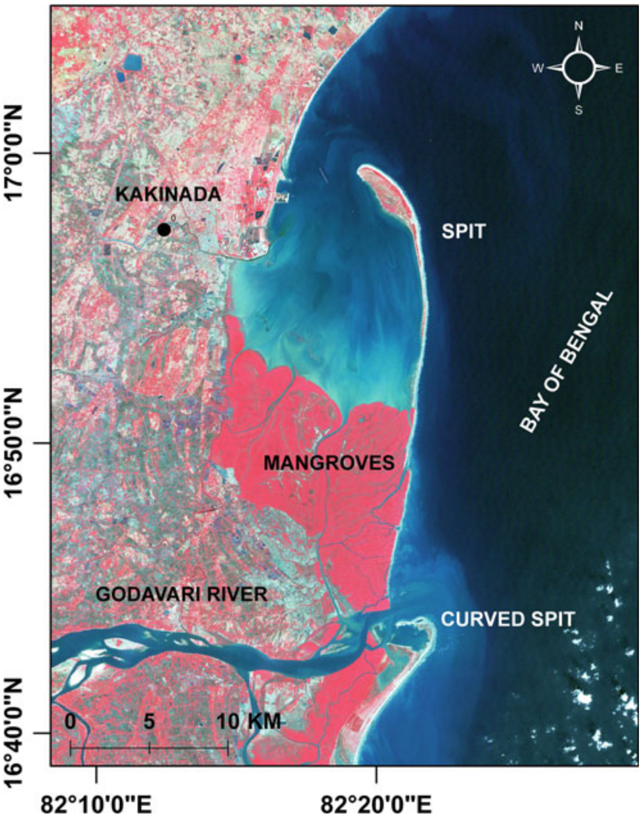
Spits are essentially narrow depositional embankment-type features that show a dominance of longshore sediment deposition over cross-shore sediment movement (Figure 10). A spit’s elongation relative to width is an

Coastal Landforms, Figure 8 Chenier plains separated laterally by other cheniers, indicating low wave energy and low gradient coasts.

Coastal Landforms, Figure 9 Beach rock at Okha, Gujarat coast, India, indicating the raised beach (Picture by P. Hanamgond).

indication of both the coastal sediment availability and net longshore-directed, wave-generated transport potential. Coastal configuration also plays an important role in the formation of spits (Carter, 1988). Spits are found on an irregular coastline where sediment availability and wave power allow a constructional smoothing of the coastline. Sand-dominated spits are the most common, whereas gravel-dominated spits occur in mid-upper latitudes, where gravel is a major component of coastal sediments. As spits are essentially a product of breaking wave activity, mud-dominated spits are unlikely to be observed. The presence of a spit generates a back-spit energy lee with low-energy currents and fine sediment stores (tidal banks and marshes).

Spits develop where wave refraction cannot accommodate the sudden changes of coastal trend and when rapid reduction in breaker approach angle reduces the longshore drift rate to zero at this point. This allows beach deposition to overshoot the directional shift in coastline. The spit builds from this depositional nucleus, and its orientation is a function of wave refraction accommodating to the changing nearshore bathymetry induced by the presence of the spit (Carter, 1988). Spits show a sequence of planform changes that are related to variation in both sediment supply and longshore transport potential and are best developed when nearshore wave approach is angled along the spit. A spit is connected to the coast and its proximal sediment source by the neck, while a spit extension occurs at the spit's distal end or terminus. A spit is usually the subaerial expression of a larger submarine feature, the distal position of which is the spit ramp. Ramp deposition controls spit growth and usually has a high fine sediment



Coastal Landforms, Figure 10 Spit on the Godavari River mouth, near Kakinada, India, as viewed by lands at satellite.

proportion related to wave-generated currents. As most of the sediment for the spit platform is supplied by longshore transport, it mimics sediment availability to the superstructure, though tending toward finer sediment (Ollerhead and Davidson-Arnott, 1995).

The spit platform requires an increasing sediment volume as the spit progressively builds into deeper water and as the volume of the superstructure generally remains the same. Spit elongation rates will decline over time if the longshore sediment supply rate does not increase. Thus, sediment supply rate is a major control on spit development. Rapid wave shoaling and landward curvature of the breaking wave crest at the spit terminus with steep bathymetric gradients lead to curvature of the distal structure against the general trend of the spit (Figure 10). High volume, but episodic, sediment supply can lead to drift-aligned spits, where the spit plan outline is essentially rectilinear despite overlapping recurves (Carter and Orford, 1991). This scenario is often associated with the initial formation of spits in a disjointed coastline, where sediment supply is formed from isolated finite sediment sources (Orford et al., 1996). Spits evolve generally in shore-parallel direction; however, inlet shoreline curvature may produce shore-normal orientations. Shore-normal spits generally have landward trends. The migration of spits into inlets influences the efficiency of an inlet to transfer water between the sea and back-barrier lagoon and generally requires a morphodynamic response by the inlet (Ortel, 1985). Spits generally retreat under rising relative sea level through overwashing and hence rollover.

Tombolo

A tombolo is a sandbar or a barrier that joins an island with a mainland or another island, resulting from longshore drift or the migration of an offshore bar toward the coast (Figure 11). Tombolos are constructive features, occurring along shorelines of submergence that are protected from large waves and where islands are common. Sediment supply is predominantly derived from the islands, yet some may also come from erosion of the shoreline, fluvial materials, underwater reefs, and offshore glacial deposits. Several types of tombolos – single, double, multiple, forked, parallel, and complex tombolos – are reflective of the coastal system (e.g., wave mechanisms) from which they are derived. Tombolos can restrict flow between the sea and intertidal zone, forming a lagoon and altering the local ecology.

Strandflats

The strandflat was first described by Reusch (1894), and its possible origin was first explained in detail by Nansen (1922). The word "strandflat" is used for the shallow sea along the western Norwegian coast and also along coasts in Arctic and Antarctic areas that have been covered by ice sheets during the Quaternary ice age. Apart from long stretches of the west coast of Norway where the strandflat is an almost continuous feature, the strandflat has also been recognized in areas as far apart as the South Shetland Isles, Alaska, and western Scotland. The low areas of strandflat often appear as broad glacially molded coastal rock platforms and backed by high cliffs (Figure 12). These shore platforms generally exhibit considerable local relief.

The processes of strandflat formation include marine abrasion, subaerial weathering, glacial erosion, frost shattering, and cold climate shore erosion. The strandflat is primarily the result of sea-ice erosion and frost shattering during the Quaternary, and the surfaces are later modified by marine and glacial erosional processes (Larsen and Holtedahl, 1985). The strandflat surfaces produced by cold climate shore processes must have been repeatedly overwhelmed by ice sheets and subject to marine processes during numerous intervals of cold climate throughout the Quaternary.

Tidal delta

Tidal deltas are large sand bodies formed within, or in the vicinity of, tidal inlets. Flood-tidal deltas form landward of the inlet mouth, under the influence of flood-tidal currents. The major morphological features of flood-tidal deltas typically include a seaward-dipping flood ramp.

Coastal Landforms, Figure 11 Tombolo on Om Beach, Gokarn, west coast of India (Picture by P. Hanamgond).

Coastal Landforms, Figure 12 Flat coastal plains, crisscrossed by glacial rivers, are known as strandflats. This view shows a 3-km-wide strandflat along the coast of Oscar II Land to the south of Engelskbukta (Source: http://www.swisseduc.ch/glaciers/arctic-islands/arctic-07-en.html?id=8).

However, landward sand movement occurs through the migration of sand waves under the action of flood currents; subtidal flood channels, which extend into the inlet and which dissect the partly intertidal landward portion of the delta; marginal ebb-aligned spits; and spillover lobes formed by the action of ebb currents over the lower parts of the ebb shield (Hayes, 1980).

Ebb-tidal deltas occur seaward of the inlet, predominantly under the influence of ebb-tidal currents and wave action. These deltas are usually comprised of an ebb channel, maintained by strong tidal currents; linear bars, formed by wave–current interactions along the margins of the ebb channel; a terminal lobe formed at the distal (seaward) end of the ebb channel, where the tidal current diminishes; and sandsheets or swash platforms formed by wave action adjacent to the ebb channel characterized by migrating swash bars.

The morphology of the tidal deltas is characterized by tidal prism, configuration of the inlet and adjacent shoreline, wave climate, and the rate of littoral sediment

Coastal Landforms, Figure 13 Tidal creeks at the southern end of Great Bay (Source: http://www.nhdfl.org/about-forests-and-lands/bureaus/natural-heritage-bureau/photo-index/Deletions/tidal-creek-bottom.aspx).

transport. In microtidal areas, flood deltas are often better developed than their ebb counterparts, owing to the dominance of landward, wave-driven, sediment transport. Ebb delta morphology is generally more variable than that of flood deltas, owing to the importance of regional and local contrasts in wave climate (Boothroyd, 1985) and due to the close coupling of delta processes with wider coastal morphodynamics. Ebb delta volume increases with the tidal prism and decreases with inlet width/depth ratio and wave energy. Under conditions of low wave energy, ebb deltas are typically more elongated and extend farther seaward.

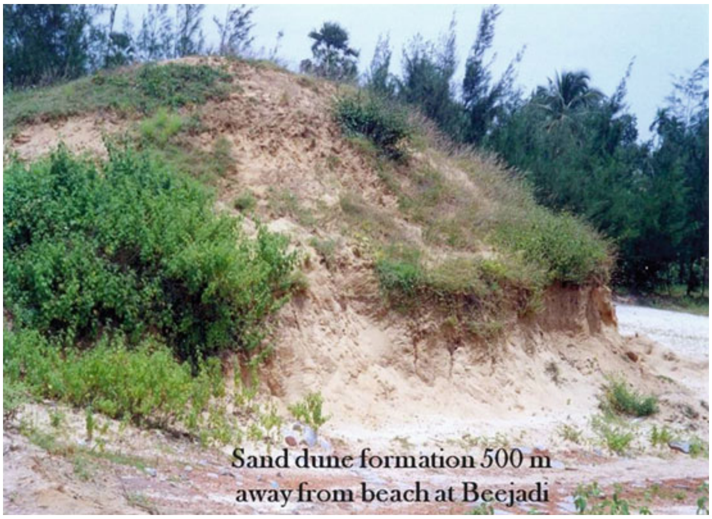
Another important landform in tidal delta/tidal flats is the tidal creek (Figure 13). Creeks occur extensively on mudflats and muddy coasts, mangrove swamps, and salt marsh surfaces (Eisma, 1998). Tidal creeks often have a high drainage density because of the large volumes of water that they drain (Pethick, 1984). The morphology of the creeks is also often distinctive. Although some may bear a superficial resemblance to dendritic river channel networks, flow along them is bidirectional (French and Stoddart, 1992). They have a tendency to taper upstream and flare downstream (Fagherazzi and Furbish, 2001), and their discharge is determined by the tidal prism. In areas with a large tidal range or rapid seaward progradation, creek systems may be markedly linear in form. In areas with cohesive sediments, creeks have steep edges, whereas in sandier areas, they tend to be shallower and wider.

Coastal dunes

Sand dunes are ubiquitous landforms along many of the world's coastlines and are indicative of periods when wind, sediment supply, vegetation, and local climate all coexisted at suitable levels to result in dune deposition (Figure 14). Once established within an accommodation space, coastal dune fields represent responsive geomorphological landscapes that react closely with changes to forcing parameters (Jackson and Cooper, 2011). Morphological behavior of dune fields is characterized by changes in climate which drives local precipitation, temperature, and wind stress over dune landforms. Sea-level rise, associated with rapid climate change scenarios, is normally tied with instability at the front edge of a sand dune coast (Carter, 1991; Saye and Pye, 2007). Under such scenario, there will be a predicted increase in vertical growth and eventual mobility of foredunes, leading to a transgressive response across the rest of the dune field (Figure 14).

Dune fields are classified in to four types – foredunes, blowout, parabolic, and transgressive dune fields. Foredunes are shore-parallel dune ridges formed on the top of the backshore by aeolian sand deposition within vegetation. They may range from scattered hummocks or nebkha, relatively flat terraces, to markedly convex ridges. Active foredunes occupy a foremost seaward position, but not all foremost dunes are foredunes. Other dune types may occupy a foremost position on eroding coasts or coasts where foredunes are unable to form. Foredunes generally fall into two main types – incipient and established foredunes.

A blowout is a saucer-, cup-, bowl-, or trough-shaped depression formed by wind erosion on a preexisting sand deposit. The adjoining accumulation of sand, the depositional lobe, derived from the depression and possibly other sources, is normally considered part of the blowout (Nordstrom et al., 1990). Blowout morphology may be highly variable, ranging from cigar-shaped, V-shaped, scooped hollow, and cauldron and corridor types, from pits to elongated notches, troughs or broad basins, and saucer and trough blowouts (Cooper, 1967). Saucer blowouts are semicircular or saucer shaped and often appear as shallow dishes. Deeper cup- or bowl-shaped blowouts



Coastal Landforms, Figure 14 Sand dunes showing the vertical growth covered with vegetation, west coast of India (Picture by K.S. Jayappa).

may evolve from them. Trough blowouts are generally more elongate, with deeper deflation floors and basins and with steeper, longer erosional lateral walls or slopes.

Parabolic dunes are typically U- and V-shaped dunes characterized by short to elongate, trailing ridges which terminate downwind in U- or V-shaped depositional lobes. The depositional lobes may be simple, relatively featureless sandsheets, or textured with a variety of dune forms (e.g., transverse dunes, barchanoidal dunes, etc.).

Transgressive dune fields are well developed on high wind and wave energy coasts with significant sediment supply and in all climatic regions. Transgressive dune fields and sheets are aeolian sand deposits formed by the downwind or alongshore movement of sand over vegetated to semi-vegetated terrain. Such sheets and dune fields may range from quite small (hundreds of meters in alongshore and landward extent) to megadune size fields. They may be completely unvegetated, partially vegetated, or fully vegetated (Nordstrom et al., 1990). Dune fields are covered with a variety of superimposed dune forms. They have also been termed mobile dunes, migratory dunes, mendano, and machair.

Coral reefs

Corals are organisms that secrete a calcareous exoskeleton and are major contributors to a coral reef. Coral reefs are natural structures of calcium carbonate made largely from the skeletons of hard corals and coralline algae. Some modern reefs have been forming for millions of years and can stretch for hundreds of kilometers off tropical coasts.

Reefs can be broadly classified as spatially heterogeneous, three-dimensional structures which have morphological form that is different from that of the underlying substrata. The term reef has been used to classify a whole host of organic and inorganic structures including stone reefs, oyster reefs, coral reefs, atolls, and algal reefs.

Reefs are found in temperate to tropical marine ecosystems, with the most prominent reef types, corals and atolls, being found in tropical and subtropical zones. Algal reefs and bioherms are commonly found in more moderate climatic zones, such as the Mediterranean. In temperate regions, reefs are often more like bioherms or biostromes in structure.

Coral reefs are found mainly between 25°N and 25°S latitudes. The reef-building (herm atypic) corals prefer sea-surface temperatures between 25 and 29 °C. Hermatypic corals mostly occur in the "photic" zone, where sufficient sunlight can penetrate for their symbiotic algae for photosynthesize. The distribution of fossil coral reefs suggests that sea-surface temperatures have constrained their spread since their appearance in the early Triassic (Birkeland, 1997). Coral reefs were alternately exposed and drowned as temperatures and sea levels oscillated during the Quaternary. During glacial periods, when sea levels were low, the distribution of coral reefs was much less and in marginal areas of the modern coral seas (like the Hawaiian Islands), the reefs died out entirely (Grigg, 1988). Owing to cooler temperatures, coral reefs grew at slower rates, and many were comparatively ephemeral. As temperatures increased and sea levels rose at the end of the glacial periods, reefs gradually became reestablished across wider areas of the seas.

Depending on oceanographic factors, upward-growing coral reefs were either able to "keep up" with rising postglacial sea level or form a drowned reef (Neumann and MacIntyre, 1985). Drowned reefs occur in many parts of the Pacific and Indian Oceans. Most of the coral reefs have failed to keep up with rising sea level associated with climate change and sea-level history, paleolatitude, seawater temperature, and light (Flood, 2001). In many parts of

Coastal Landforms, Figure 15 Photograph showing the 8-km-wide Atafu Atoll located in the southern Pacific Ocean (Source: http://earthobservatory.nasa.gov/IOTD/view.php?id=37753).

the world, coral reefs are found raised above their modern counterparts and, as such, often provide important insights into reef structure and history.

The morphology and genesis of coral reefs vary significantly. On the basis of their form, reefs may be divided into atoll reefs, barrier reefs, and fringing reefs (Nunn, 1994). Barrier and atoll reefs are older, often being composed of reefs of many different ages; reef upgrowth during postglacial periods has been followed by subaerial exposure, and erosion during the glacial periods followed by renewed upgrowth.

Fringing reefs are juvenile and the youngest and most ephemeral of the three forms, and they grow outwards from a coast. They are located close to the land and indeed cannot exist very far away from the land. Unlike atoll reefs and barrier reefs, most fringing reefs are formed as discrete units during the recent period of postglacial sea-level rise. Most of them began growing from shallow depths on the flanks of a tropical coastline when ocean-water temperatures at the end of the glacial period became suitable for reef growth (Neumann and MacIntyre, 1985). Fringing reefs are mostly affected by humans.

Atolls

Coral atolls, dispersed widely throughout the warm waters of the tropical Pacific and Indian Oceans, are among the world's most impressive biogenic landforms (Nunn, 2010). The classic exposition of atoll origin was first explained by Charles Darwin in 1836, where he observed a barrier reef surrounding Mo'orea Island during his visit to the Keeling Islands. Darwin set out his theory of atoll development which involved the upward growth of a coral reef in response to the subsidence of its foundations (Darwin, 1842). Darwin's elegant theory was founded on the premise of a subsiding volcanic island and the corresponding upward growth of friging and barrier coral reefs keeping pace with the rising relative sea levels (Terry and Goff, 2013). Darwin suggested that it was the tendency of ancient volcanic islands in the oceans to subside, but their coral fringe could stay alive only if it was able to grow upward at the same rate. Thus, modern atoll reefs are only veneers of living coral growing atop a coral framework composed largely of the skeletal remains of dead hermatypic (reef-building) corals.

Atolls are generally subcircular rings of coral reef (Figure 15) surrounding a lagoon with no dry land other than occasional islands (called *motu*) made from sand and gravel-sized detritus thrown up on the reef during storms (Nunn, 1994). The word "atoll" should be applied only to the reef, but sometimes the term is used more loosely to refer to *motu*. In the Pacific, where some of the world's oldest atolls exist, many have reef foundations dating from at least the Oligocene. It is a surprise to know how such organic structures remain intact despite the continuous buffering of storm waves, earthquakes, and even nuclear weapons tests. Johnston Atoll in the central Pacific, where the US chemical weapon stocks are being destroyed, lost its southern flank in a series of huge landslides predating its discovery by humans. On the other hand, part of Moruroa Atoll in French Polynesia, where 98 subterranean tests of nuclear bombs were carried out between 1981 and 1991, has subsided as a direct consequence of nuclear tests (Keating, 1998). Many atolls exhibit major arcuate "bight-like" structures in their plan form. These departures from circular or elliptical forms are indicative of geomorphological processes that cannot be ignored (Terry and Goff, 2013).

Summary

The coastal landforms, carved out during the Late Quaternary period, are influenced and modified primarily by sea-level changes, ocean and river-mouth processes, sediment nature and its supply, and human activity. The important depositional coastal landforms are barrier islands, coastal bars, lagoonal systems, and mudflats that are characterized mostly by tides, waves, currents, and grain size, as well as beaches and spits that occur in all tide and wave conditions. In the formation of spits, wave refraction and longshore drift, apart from coastal configuration, play an important role. Beach ridges and chenier plains are indicators of past sea-level changes, climatic conditions, wave regime, and sediment supply. Coastal dunes represent geomorphological landscapes that are reworked by climate drivers such as local precipitation, temperature, and wind. Transgressive dune fields indicate high wind and wave energy coasts with large sediment supply.

Coral reefs mainly confine to 25°N – 25°S latitudes, and their occurrence suggests that sea-surface temperatures play an important role in their distribution. Based on the morphology, reefs are classified into atoll, barrier, and fringe reefs. Darwin (1842) was the first who proposed a theory on evolution of coral reefs, and he hypothesized that subsiding volcanic islands correspond with upward growth of fringing and barrier coral reefs keeping pace with relative sea-level rise.

The impact of human activities is the primary concern for the fragile nature of coastal landforms and coastal ecosystems.

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Cross-references

Back Dune
Bar
Climate Change
Coastal Barriers
Coastal Bays
Coastal Lagoons
Deltas
Estuarine Beaches
Foredune
Mangroves
Saltmarshes
Secondary Dune
Spit
Tidal Flat
Tides

COASTAL RISKS: FLOODS

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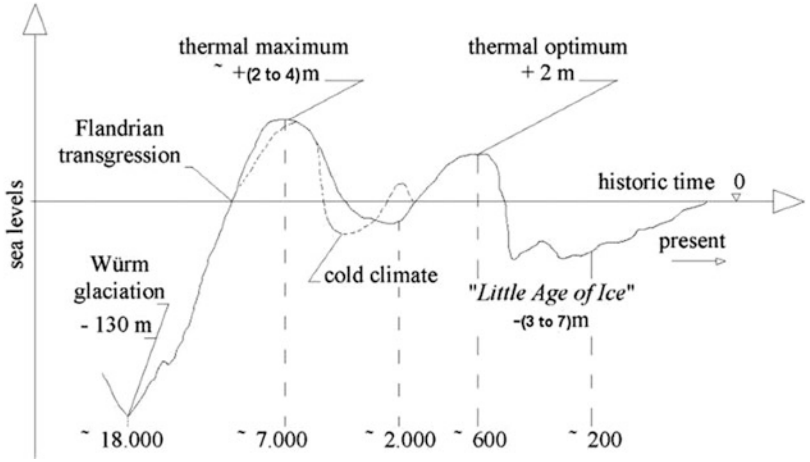
Definition

Flooding of coastal lands is primarily due to inundation by the sea during storms and other natural events (e.g., tsunamis) that increase coastal population risk. Flooding also occurs from inland waters when storm water levels hinder evacuation.

Description

Coastal risks have very different origins and etiologies, but current vulnerability assessments focus mainly on erosion and climate-induced floods. Generalized fluid-dynamic erosion is the reason for most shore and coastal protection measures. Tectonic plate movements may generate tsunami waves, and the resulting coastal risks can reach great levels for vulnerable settlements. Hurricanes and regional monsoons can also cause vast erosion and damage to coastal zones.

Eustatic sea level is increasing due to climate change (Figure 1), but crustal isostasy, tectonics, and coastal plain subsidence cause variability of sea level at the local level. Accordingly, this trend of increasing global sea level can be accentuated in some places or attenuated and even reversed in others. The intensity and frequency of coastal flooding depend not only on eustasy but also on other climate-related factors such as low-pressure systems and strong winds that can raise the average sea level above the current tide and generate temporary increases in basin water levels that cause inland coastal floods (Diez et al., 2011, 2012). A temporary rise in sea level can act as a dam at the mouth of a river causing blockage of river drainage and a rise in water levels on the river, on its



Coastal Risks: Floods, Figure 1 Global Climate Evolution since last Glaciation expressed on mean sea level and based on geo-historic approaches. Scales vary decreasing to present time and level.

floodplain, and on surrounding areas (Audiencia Territorial de Valencia, 1991).

Unlike marine floods that are caused by seawater in limited areas, coastal floods may be due to a variety of different causes, and they may affect much larger coastal zones. When such floods are directly sourced to pluvial or fluvial waters, persistent high sea level is the main cause of the flood duration.

Historically, structural coasts were selected for settlements mainly because of security, health, safety, and economic reasons. However, sedimentary coasts offered greater productivity, and these areas soon attracted large populations to their plain littoral/deltaic hillocks. Mediterranean seaside ("maritime") cities were always located beside a castled rock-hill (e.g., Athens, Haifa, Genoa, Malaga, and Monaco), whereas coastal plain cities were established on the landward side of relatively dry and high grounds (e.g., Rome on its seven hills, Valencia or Alexandria on delta hillocks, Venice on a relict barrier island of the Po Delta, etc.).

Coastal cities and towns often spread into lowland areas. Enormous conurbations are at a huge risk of flooding today as a result. The case of New Orleans is paradigmatic: its older settlements barely suffered from floods caused by Hurricane Katrina, meanwhile most of its later developments and lower quarters were catastrophically flooded. The great European delta formed at the outlet of the Elm, Rhine, and Meuse rivers has required drastic transformations to protect large cities from flood risk.

Flood risk can never be totally eradicated. Therefore, each vulnerable coastal development now requires a risk management plan to deal with the hazard. Flood risk has become a datum for analysis and resiliency management.

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Cross-references

Climate Change
Coastal Erosion Control
Shoreline Changes
Submerged Coasts
Submergent Shoreline
Uplifted Coasts

COASTAL SQUEEZE

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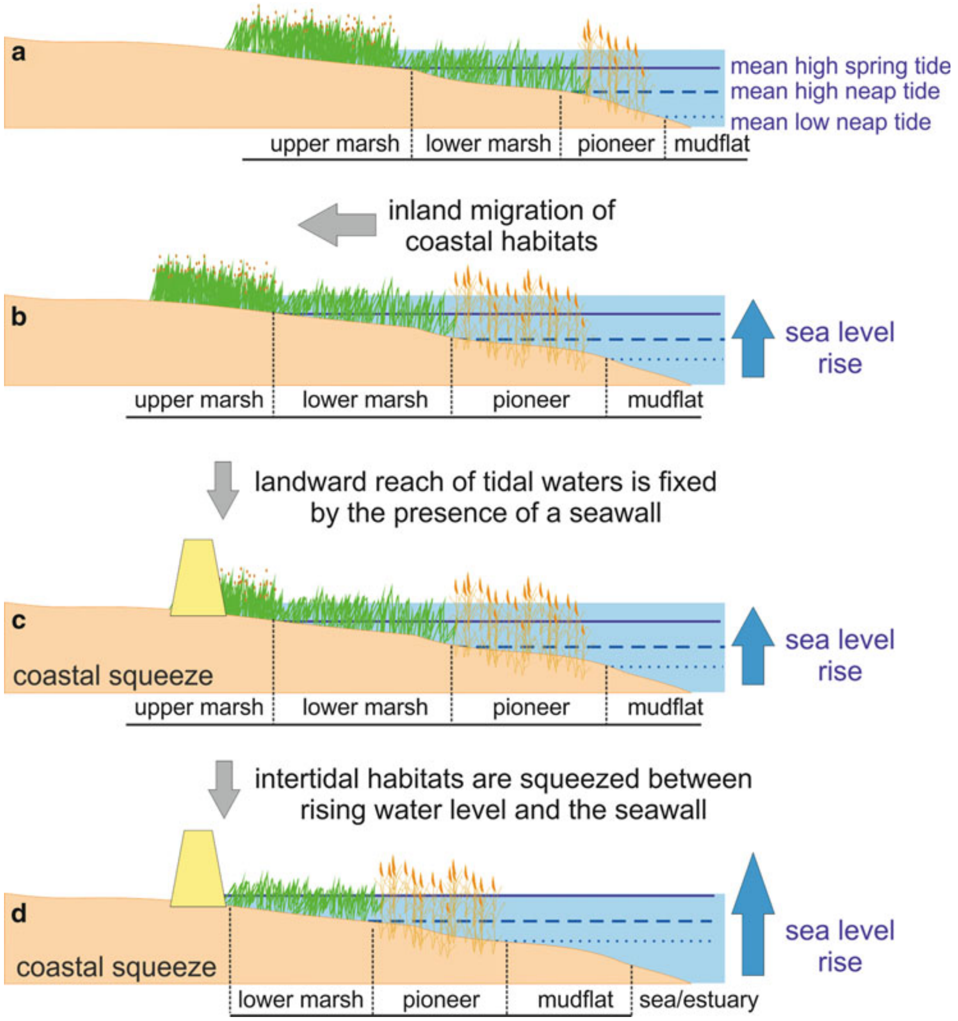
Definition

Coastal squeeze refers to the loss of intertidal habitats due to rising sea levels along coastlines fixed by hard engineering structures. The term coastal squeeze should not be used to refer to losses due to natural processes (Pontee, 2013).

Natural coasts can dynamically adjust to changing meteorological and climatic conditions. In natural systems, rising sea levels usually result in a landward movement of habitats (Figure 1a, b). Salt marshes, for example, depending on a number of interacting physical and biotic variables, can migrate inland and accrete vertically, naturally adjusting to sea-level rise. The natural



Coastal Squeeze, Figure 1 The elevation in relation to the tidal range is one of the key factors determining the type of intertidal habitat that may develop in a particular location (**a**). Natural habitats tend to migrate inland as a response to rising sea levels (**b**). As a result of this migration the intertidal area may expand or reduce depending, for example, on the coastal topography. Hard engineering structures will invariably fix the landward limit of intertidal areas (**c**), which will be reduced in extent as sea levels rise and more land becomes permanently inundated (**d**). The loss of coastal habitats due to rising sea levels in front of artificially fixed shorelines is known as coastal squeeze.

landward migration of habitats is prevented in coastlines “fixed” by hard coastal engineering, leading to coastal squeeze (French, 1997).

The type of intertidal wetland that may be established at any particular location is influenced (among other variables) by their position within the tidal range (Figure 1a). The vertical zonation of marshes reflects the tolerance of species to inundation (Pennings and Calloway, 1992), i.e., more tolerant species are found at lower elevations. Coastal defences fix the upper boundary of intertidal habitats (Figure 1c, d); therefore, a rise in sea level will gradually increase the frequency and duration of inundation and ultimately result in loss of intertidal area (as lower areas become permanently submerged). Depending on the range of elevations in relation to the water levels, increased exposure to inundation may lead to a shift in the types of marsh communities and/or the loss of habitats. Mudflats may occupy areas formerly dominated by pioneer marshes (Figure 1d); these might shift to higher ground or will disappear if suitable conditions are not available. The same process applies to other types of marshes.

Coastal squeeze and land reclamation are often cited as the main causes for the loss of intertidal habitats (e.g., Doody, 2012). Coastal squeeze is not the only cause for the loss of intertidal habitats. Hughes and Paramor (2004) argue that coastal squeeze would lead first to the loss of upper marshes, while the loss of pioneer marshes is most commonly observed. The authors suggest that increases in the abundance of the polychaete *Nereis* might be the cause of widespread loss of pioneer marshes in southeast England. The impact of storms along the coast of the Gulf of Mexico has been identified as one of the main reasons for the increased rate of wetland loss in the United States in the period 2004-2009 when compared with the previous five years (Dahl and Stedman, 2013). The loss of salt marshes is particularly concerning as they provide natural coastal protection and other valuable ecosystems services.

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COASTAL WETLANDS

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Definition

Coastal wetlands are habitats in close proximity to oceanic or estuarine waters that are directly influenced by tides and are covered permanently or periodically with salt or brackish water. In the broad sense, they include submerged seagrass communities, tidal flats, and emerged salt marshes and mangrove forests.

Coastal wetlands

Tidal flats are ecotonal areas between land and sea and can extend from the subtidal through the intertidal and into the supratidal zones. Although they often appear barren, they can be highly productive and support large animal populations. They occur throughout the world in areas with significant fine-grained sediment deposition. Seagrasses are submerged flowering plants occurring in protected shallow estuaries generally with soft sediments. They can form extensive beds that are important habitat for a large number of animal species and play an important ecological role in the nearshore estuarine environment. See Tidal Flat for more details. Below we consider emergent salt marshes and mangroves.

We distinguish between coastal wetlands (situated at or near the coast with direct influence of seawater salinity) and tidal wetlands which can include freshwater areas a considerable distance from the ocean but whose hydrology is still influenced by tidal phenomena that can propagate substantial distances upland (Rey et al., 2012a). More or less distinct vegetation zones dictated principally by tidal inundation are common in coastal wetlands, but vegetation mosaics and mixes are also widespread, and sharp transitions often occur in response to slight environmental gradients (Marani et al., 2013).

The structure and function of coastal wetlands are determined by many interconnected processes and feedback loops that operate at varying temporal and spatial scales (Berger et al., 2008). Top-down constraints such as landform and climate interact with bottom-up effects such as local competition and individual plant photosynthesis to affect the structure and function of a given wetland. Twilley and Rivera-Monroy (2005) divided these processes into three types: *regulators* (non-resource factors such as salinity, climate, etc.), *resources* (factors used by organisms for growth such as nutrients, sunlight, and space), and *hydroperiod* (the duration, frequency, and depth of flooding).

The hydrological pattern of coastal wetlands is the dominant factor affecting their structure and function (Mitsch and Gosselink, 2007). Hydrology affects many biotic and abiotic processes which in turn may modify hydrology. Examples of these include primary and secondary productivity, soil and water chemistry including anaerobiosis, nutrient cycling, salinity, biological diversity, carbon cycling, sedimentation dynamics, and microbial metabolism. In addition to climate and basin morphology, tidal flooding and flow through tidal creeks and channels (Perillo, 2009) are often the most important components of coastal wetland hydrological dynamics. Other important factors include subsurface composition, precipitation, surface flows, ground water flows, and evapotranspiration. Sedimentation dynamics, which includes production, transport, and sediment storage and is heavily influenced by hydrology, also plays a critical role in wetland function and maintenance (D'Alpaos et al., 2012).

The modification and transport of chemicals through coastal wetland ecosystems (biogeochemical cycling) result from a complex matrix of chemical, physical, and biological processes, again, with numerous feedback mechanisms, and give rise to many of the well-known wetland functions such as carbon sequestration/export, nutrient exports, and many others. Biogeochemical cycling interacts with marsh hydrology and geomorphology to determine physical and biological conditions within a given wetland. Major chemical cycles in coastal wetlands include those of nitrogen, sulfur, iron, manganese, carbon, and phosphorous.

The structure and function of wetland biological communities are closely tied to and have a heavy influence on other wetland processes (Marani et al., 2007). Primary production by marsh vegetation directly or indirectly influences a variety of food webs including wetlands, estuarine, oceanic, and terrestrial-based ones. Vegetation, together with wetland physiographic features such as tidal creeks and elevation discontinuities, provides critical habitat for a wide variety of terrestrial and aquatic organisms. Ecological interactions such as competition and predation are also crucial in modifying community structure. Belowground bacterial activity has important effects upon biogeochemical cycles. Wetland chemistry and hydrology directly influence plant and animal communities, but biological activity, for example, the burrowing of fiddler crabs (McCraith et al., 2003) or emergent plant metabolism (Gribsholt et al., 2003; Gribsholt and Kristensen, 2003), can have important consequences for wetland hydrology and chemistry.

Importance of coastal wetlands

Coastal wetlands have great ecological importance because of their biodiversity (Gopal and Junk, 2013) and productivity. They are often critical habitats for protected and endangered species and for species of commercial or recreational fishery value. Wetlands are integral components of coastal hydrological processes and function in flood control and in retention, transport, and storage of carbon, sediments, nutrients, and pollutants. Wetlands often filter contaminants originating higher up in the watershed and act as sinks for excess nutrients, thus contributing to the maintenance of estuarine and nearshore oceanic water quality. Wetlands often provide erosion protection and sediment stabilization. Wetlands also have high recreational and aesthetic values that make these areas desirable for human habitation. Additionally, the coastal zone in general is highly valuable economically, with many important facilities such as ports and airports and the industrial/commercial development that they attract often situated there. As a result, more than 40 % of the US population lives in coastal counties (NOAA, 2013), and over 44 % of the world's population lives within 150 km of the coast (UN, 2013).

General distribution of coastal wetlands

Below is a very general outline of coastal wetland distribution throughout the world. It only offers broad descriptions, and individual localized sites may depart significantly from regional norms.

Polar coastal wetlands

High-latitude coastal wetlands consist of salt and brackish water marshes and laida (wetlands inundated by both salt water during storms and freshwater during snowmelt). They occur along most coasts in the Northern Hemisphere, with some of the more extensive ones occurring along Hudson and James Bays and along the coastal plains of Alaska and the Yukon (Martini et al., 2013). Large wetlands also occur along the Russian coast and in major river deltas of the region. Similar wetlands do not occur in the Southern Hemisphere because of a dearth of ice-free substrate (Martini et al., 2013).

North America

Farther south, between New Brunswick and Nova Scotia in the Gulf of Maine, the Bay of Fundy wetlands occur at the approximate subarctic-temperate transition zone. This area has one of the largest tides in the world, with spring tidal ranges of close to 15 m. Extensive low marshes are populated almost exclusively by *S. alterniflora*, whereas in the high marsh *S. patens* is most widespread and *Phragmites australis and Iva frutescens* occur along the upper edge. Within New England, regional differences associated with climate and human impacts exist, but in general, marshes standing on marine peat have similar vegetative composition as above. *Juncus gerardii* is common as an upland fringe, and in areas with substantial freshwater inputs, a brackish community consisting of *Scirpus americanus*, several *Typha* species, *Zizania aquatica*, and *Phragmites australis* occurs (Nixon and Oviatt, 1973; Pratolongo et al., 2013).

From New England south to northern Florida, marshes develop behind protective barrier island complexes. Throughout this area, *S. alterniflora* dominates the low marsh, with a tall form occurring in areas with longer flooding periods and a short form where daily tidal inundation lasts only for a short time. In the high marsh, *S. patens* or *Juncus roemerianus* may form monospecific stands or the two species may codominate, often forming complex spatial patchworks. As in New England, transitional brackish areas exist which in this case are dominated by *Spartina cynosuroides*.

In Florida, south of 30°N latitude, mangroves gradually replace salt marshes, but narrow bands of salt marsh can be found throughout the state. Three mangrove species occur in Florida: the red mangrove (*Rhizophora mangle*), the black mangrove (*Avicennia germinans*), and the white mangrove (*Laguncularia racemosa*). A variety of herbaceous halophytes often occupy the mangrove understory; examples include *S. alterniflora*, *Batis maritima*, and *Salicornia virginica*.

East of the Mississippi deltaic wetlands, along the northern Gulf Coast of North America, grass/rush marshes can be found usually directly in front of the open ocean due to the low tidal energy in the region. Clearly delimited low marsh and high marsh plant zones are often evident, but convoluted plant community mixtures can be just as common (Montague and Wiegert, 1990). Generally, *S. alterniflora* forms relatively narrow bands along the shoreline and is then replaced by black needle rush (*Juncus roemerianus*), but both species can also form extensive monocultures (Kurz and Wagner, 1957; Rey et al., 2012a).

The Mississippi deltaic plain region supports extensive wetland complexes (approximately 7,250 km^2) in six major drainage basins that represent a time series of shifts in the major channel of the river. The youngest basins support mostly freshwater marshes because of their shallow depth and the large volume of freshwater inflow. The older basins support a variety of wetland types including extensive salt marshes and forested wetlands. West of the Mississippi, coastal wetlands tend to occur along protected shores behind barrier islands and along protected bays, in once flooded ancient river valleys.

Along the Pacific coast of North America, coastal wetlands are sparse because of the rugged terrain. Small isolated wetlands along river valleys are dominated by *Spartina foliosa* (a species that also exhibits short and tall forms) accompanied by various succulents. Farther north, the San Francisco Bay area supports wetlands dominated by *S. foliosa* in the low marsh and *S. virginica*, *Jaumea carnosa*, *Triglochin maritima*, and *D. spicata* in the high marsh. In the Pacific Northwest, low marshes are dominated by halophytic succulents such as *Salicornia virginica*, *Jaumea carnosa*, and *Triglochin maritima* as well as several grasses and sedges such as *Distichlis spicata* and *Carex lyngbyei*. Tufted hair grass (*Deschampsia cespitosa*) commonly dominates in high marshes, accompanied by mixes of many other species.

Central America and Caribbean

In the Caribbean islands, mangroves predominate as fringe vegetation along the coast, in protected bays and lagoons, and as overwash islands that are often completely flooded by each tidal cycle. In Central America, more extensive mangals develop, and herbaceous marshes can be found as small isolated pockets or as narrow fringes in front of mangrove formations.

South America

As in North America, only small isolated wetlands can be found along the Pacific coast because of the rugged terrain associated with the Andean chain. Extensive wetlands occur on the Atlantic coast. Mangroves can be found throughout the northern part of the continent, with 90 % of the coverage by approximately 10 South American species found in Brazil, Colombia, Venezuela, Ecuador, and Suriname (FAO, 2007). Typical salt marshes occur farther south and are dominated by *S. alterniflora* in the low marsh and *S. densiflora* in the high marsh along with associates such as *Limonium brasiliense*, *Juncus acutus*, and *Distichlis spicata*. As one approaches Tierra del Fuego, the region becomes arid and cold and vegetation everywhere becomes scant.

Europe

Northern European marshes are characterized by a pioneer zone thinly vegetated with *Spartina anglica* and *Salicornia* spp., an intermediate zone populated with a variety of halophytes such as *Aster tripolium*, and an upper marsh zone where *Festuca rubra*, *Juncus gerardii*, and *Elymus athericus* predominate. Recently, the latter species has been invading the mid and low marsh areas (Pratolongo et al., 2013). Large salt marshes exist along the Atlantic coast and along the North, Baltic, and Walden seas. Coastal wetlands in the Mediterranean region are commonly associated with river mouths; important deltaic wetlands include the Camargue (Rhone) in France and the Ebro Delta (Ebro) in Spain. In this area uncharacteristically, shrubby species such as *Sarcocornia fruticosa* predominate in the low marsh and *Limonium* spp. in the mid-marsh, and various *Juncus* species usually populate the high marsh.

Asia

The variety of climatic and edaphic conditions, the extent of the coastline, and the high frequency of embayments, islands, flats, estuaries, and river deltas, particularly in S.E. Asia, result in the highest biodiversity and the greatest areal coverage of mangroves in the world. Over 55 species of true mangroves occur in Asia (FAO, 2007). In East Asia, extensive coastal marshes occur along deltas formed by major rivers such as the Yangtze and Huang He. *S. alterniflora* was introduced to several areas in China in the late 1970s and has displaced many of the native plants, particularly in the lower marsh areas (Lu and Zhang, 2013).

Oceania

This region, which includes Australia, Papua New Guinea, New Zealand, and the S. Pacific islands, has a very high mangrove biodiversity (close to 50 species) but low areal extent of mangroves. They are found in protected bays, estuaries, lagoons, and coral atolls in the region. Close to 75 % of the mangrove coverage is concentrated in Australia (FAO, 2007). In Australia, *Spartina anglica* can occur seaward of mangrove areas. In more temperate areas, more conventional salt marsh zonation occurs.

Africa

Mangroves occur throughout the continent. On the east coast, mangroves often occur as narrow fringes except where large deltas (e.g., in Mozambique and Tanzania) allow the development of more extensive forests. Much broader forests develop in the west coast, culminating in the vast mangal associated with the Niger River delta that extends up to 40 km inland and supports very large trees that can reach 40 m in height. Salt marshes occur on the Mediterranean, Red Sea, and Indian Ocean coasts often behind mangroves. In Southern Africa, mangroves occur as far south as Angola, and extensive salt marshes south of there (Hughes et al., 1992).

Impacts to coastal wetlands

Human pressures currently cause the greatest impacts to wetlands. These include outright habitat loss due to residential, industrial, and agricultural development and associated infrastructure; or habitat degradation as a result of pollution, hydrological changes, and other impacts from surrounding human activities. Pest control activities mostly for mosquitoes can impact coastal wetlands, but habitat management for mosquito control has also been used as a marsh restoration tool (Rey et al., 2012b). In the United States, there were wetland losses of 146,200 ha in coastal watersheds of the eastern seacoast between 1998 and 2004 in spite of overall gains in wetland coverage during the same period (Stedman and Dahl, 2008). Between 2004 and 2009, salt marsh and estuarine emergent areas declined by 45,140 ha (Dahl, 2011). Worldwide mangrove losses between 1980 and 2005 have been estimated at more than 3.5 million ha (FAO, 2007) but the actual losses may be significantly greater (Giri et al., 2011).

As habitats that bridge marine and terrestrial ecosystems, coastal wetlands are particularly vulnerable to sea level changes and increased frequency of storms and other extreme events produced by climate change (Hopkinson et al., 2008). Depending upon circumstances, coastal wetlands may keep up with the relative rise, be lost, be degraded, or migrate landward in response to sea level rise (Gilman et al., 2007).

Summary

Coastal wetlands include seagrass communities, tidal flats, coastal salt marshes, and mangrove forests. They are important and complex ecosystems whose structure and function are determined by a large number of biotic and abiotic processes including non-resource factors such as salinity and climate, resources used by organisms for growth, and hydroperiod, with the latter being the dominant factor in salt marshes. These areas have great ecological importance due to the value and diversity of ecosystem functions that they provide. However, because of the desirability of the coastal zone for human habitation and associated infrastructure, coastal habitat degradation and loss is a serious problem worldwide. Also, because of their location in the interphase between the sea and the land, these habitats are particularly vulnerable to sea level changes and increased frequency of storms and other extreme events produced by climate change.

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Cross-references

Barrier Island
Cordgrass
Deltas
Estuarine Deltaic Wetlands
Fiddler Crabs
Food Web/Trophic Dynamics
Mangroves
Nutrient Dynamics
Saltmarshes
Sandflat
Sea-Level Change and Coastal Wetlands
Tidal Flat
Wetlands

CORDGRASS

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Synonyms

Salt-marsh cordgrass; *Spartina* spp.

Definition

Cordgrass refers to species within the *Spartina* genus of grasses (Poaceae). Cordgrass is the dominant plant in salt marshes in many regions.

Description

The genus *Spartina* includes 15 species (ITIS, 2013). Cordgrass species have long slender leaves and tall inflorescences with many spikelets, each holding many seeds. They are perennial, wind pollinated, and, often, dispersal limited.

Cordgrass species are halophytic, with special adaptations like salt-excreting glands and aerenchymous rhizomes for saline and anoxic soil conditions. They are often the dominant plant in edaphically stressful intertidal salt-marsh ecosystems. In western Atlantic and Gulf of Mexico salt marshes, *Spartina alterniflora* is the dominant plant species, confined to the low marsh by competition with its congener, *S. patens*, which dominates the high marsh (Bertness, 1991). *S. densiflora* is the dominant plant in South American salt marshes.

Cordgrass species are highly invasive and often able to hybridize with each other. *S. townsendii* and *S. anglica* species originated by hybridization within the last 150 years (Daehler and Strong, 1996). *S. densiflora* is invasive in Spain (Nieva et al., 2005) and in California, where it has hybridized with the native *S. foliosa* (Ayres et al., 2008). *S. anglica*, *S. alterniflora*, and *S. patens* have been introduced in China (An et al., 2007) and in the western United States (Daehler and Strong, 1996).

Cordgrass has been introduced intentionally for shoreline stabilization (Ranwell, 1967). Cordgrass is a marsh builder and a true ecosystem engineer (Gedan et al., 2011). Cordgrass is also valuable as livestock fodder, central to salt-marsh food webs, and efficient at sequestering carbon and nitrogen.

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Cross-references

Coastal Wetlands
Estuarine Deltaic Wetlands
Marsh Islands
Salt Marsh Accretion
Saltmarshes
Wetlands

CULTURAL SEAFOOD MANAGEMENT

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Definition

Cultural seafood management is seafood management based on customary (traditional) fishing rights and the integration of traditional knowledge in estuarine and coastal management.

Introduction

In most countries artisanal fisheries have existed for centuries with fishers using traditional and local knowledge in community or citizen-based management systems. While these systems are usually small scale, there is increasing interest in expanding or integrating these systems into larger and more complex systems (Reis and D'Incao, 2000). This need is driven by the knowledge that these habitats have changed and that seafood resources are limited. Increasingly there is recognition that traditional ecological knowledge (TEK) held by indigenous communities can play a valuable role in the management of natural resources including fisheries (Mathew, 2011). Practical skills, wisdom, and knowledge accumulated over successive generations can contribute to knowledge about species, their distributions, life histories, and behavior (Butler et al., 2012).

Throughout the world, estuaries are used for obtaining seafood and, when traditional knowledge is used with Western science and management knowledge (SMK), there are opportunities for generating a diversity of information for problem solving. Inshore marine fisheries such as those found in estuaries are often small; however, they may involve multiple species that are taken for subsistence. For example, in the islands of Torres Strait between Queensland and Papua New Guinea, people have high consumption of seafood based on up to 350 species, one of the highest numbers per capita in the world.

There are various levels of governance for seafood resources, and increasingly these are shared with government agencies. Management systems are generally region specific and may be driven by cultural keystone species. Cultural values, co-management, and power sharing of estuarine resources within a legislative framework occur in many countries, the first nation groups in North America, Melanesia, northern Australia, and New Zealand. The focus here is to use selective examples to illustrate the use of integrated management systems and their effectiveness in protecting seafood resources for future generations.

Traditional Ecological Knowledge (TEK) and Western Science and Management Knowledge (SMK)

Indigenous knowledge is used as a general term when describing knowledge systems of indigenous peoples, whereas traditional ecological knowledge is defined by Berkes (2003) as "a cumulative body of knowledge, practice and belief, evolving by adaptive processes and handed down through generations by cultural transmission." In a recent review, Thornton and Scheer (2012) use the term local and traditional ecological knowledge (LTK) which is described as a body of knowledge and not just a collection of facts. This knowledge is highly valued and has been used politically and strategically for territorial claims in many places including Oceania, North America, Central America, South Asia, and Southeast Asia. TEK/LTK is a knowledge-practice-belief complex, which is operated in many small fishing communities and where the fishery depends mainly on local knowledge of species rather than management systems, institutions, or world views. Knowledge can be grouped into nine cultural domains, as used by the Torres Strait islanders who occupy more than 150 islands between Queensland, Australia, and Western Province, PNG (Smyth et al., 2006, cited in Butler et al., 2012). Cultural domains include food source and skills associated with gathering, how it is used, including trade, environmental knowledge associated with collecting the food and "totem," the connection to various groups and mythological aspects, ceremony, beliefs, and art.

In contrast, the Western science management knowledge base (SMK) is usually underpinned by federal, state laws, and regulations implemented by agencies and departments. In most countries the responsibility for the marine and estuarine environment is split between multiple government agencies (with responsibilities defined by law) and operated by defense forces, coast guards, wildlife agencies, environmental protection agencies, health departments, fisheries, local authorities, and others. The areas under their control are large, and the managements driven by economic gains.

There are obvious differences in the form of the knowledge base resulting from SMK, which consists of published reports, qualitative analyses, and sharing of information, compared with TEK, where the information is transmitted orally and often only within families. Jokiel et al. (2011) compared major aspects of the traditional and Western knowledge systems for inshore reefs in Hawaii and listed other major differences including authority, enforcement, and resource monitoring.

Cultural keystone species

Ecologists recognize that some species have a key role in the structure and functioning of ecosystems and are essential in maintaining ecological balance. Similarly, in many societies, plants and animals can shape the cultural identity of a people. Several species together play a social role by interacting with each other forming a cultural grouping similar to a keystone guild. Cultural keystone species include green turtles and dugong for fishers from developing regions of Melanesia (Garibaldi and Turner, 2004). Their importance is reflected in the fundamental roles these species play in diet, materials, medicine, and/or spiritual practices. In Torres Strait, these keystone species have been important in creating links and understanding between TEK and SMK. The success of these is because of their cross-cultural values for the islanders and government conservation directives for the species. Another spin-off from the arrangement includes the establishment of a ranger program allowing community-based management of invertebrate species such as trochus and beche de la mer. Some estuarine cultural keystone species are large and iconic marine species, which have extensive ranges into coastal waters and provide ecosystem services and benefit from national as well as international beneficiaries. This has resulted in cross-scale partnerships between multiple indigenous communities and state and national government agencies. In broadening the approach from small-scale management to larger areas, several studies have reported successes in conservation and restoration by recognizing and focusing on cultural keystone species.

Traditional fisheries management

In many parts of the world, estuarine systems are threatened; anthropogenic changes have caused changes to estuarine habitats and their ecology. Furthermore, demographic changes, social political pressures, urbanization, education, commercialization, and technological advances have led to changed perceptions about the value of marine resources. Traditional community-based fisheries are usually small scale. Fishing is an integral part of many estuarine communities with immense cultural significance including beliefs about their origins and traditions. Ruddle (1993) provided numerous examples of traditional management from the Asia-Pacific region, including examples which he regarded as unsuccessful and those that were successful. He suggested several ways forward, favoring the use of legislation to reinforce but specify the power of traditional rights. In South Asia, one community-based traditional management system is Padu, which is based on caste and gender, a managed prawn or shrimp fishery in local lagoons where fishermen catch shrimp as they migrate from the estuary back to the sea. In recognizing the pressures exerted on management systems, previously unlicensed fishers from Kerala, Southern India, challenged the decision to prevent them from fishing in traditional areas and in the municipal courts gained official access to the fishery. The groups were well organized and charged with facilitating equitable access, providing social responsibility and providing mechanisms for rule making and resolving conflicts (Lobe and Berkes, 2004). More recently, Coulthard (2011) analyzed the Padu system in the Pulicat lagoon, India, where there are more than 30,000 artisan fishers. Here she describes the system as being in a fragile state due to poverty and reduced income because of widely fluctuating catches. Still the fishers remain loyal to the Padu system which provides political power, social standing, and prestige. This system is quite different to the systems used by larger commercial fisheries which run alongside the traditional fisheries usually with science-based stock assessments for individual species. They use fishery-specific tools (not discussed here) such as licensing, temporal closures, gear restrictions, and size limits. In the future it seems likely that such intervention measures will need to be included in traditional fisheries so that they can be of benefit to future generations.

Natural resource management

For centuries the Polynesians, who inhabited Hawaii, used a management system "ahupua'a," an integrated watershed management system between freshwater and the nearby marine coastal environment, based on ecosystem linkages between the mountains and the sea (Jokiel et al., 2011). This concept gained prominence in New Zealand, where centuries of observation and the continued practice of gathering mahinga kai and kaimoana, Ngāi Tahu whāui have built a unique body of experience and knowledge which is important for understanding the environment and maintaining its health and well-being. Mauri, which is an important part of Māori culture, is the energy or life force which is sacred and a spiritual link to the past the present and the future. Pauling (2003) developed cultural tools for mountains to the sea natural resource monitoring, including evaluating mahinga kai (food) and resources. The state of the Takiwā assessments, Te Āhuatanga o te Ihutai, was undertaken to establish the cultural health of the Avon-Heathcote Ihutai Estuary, a small estuary near Christchurch, in the South Island of New Zealand, where modifications of the estuary and the previous release of human waste made the estuary unsafe and culturally unacceptable for gathering kai moana. Surveys were undertaken to provide an assessment of the current health and provide ideas about how future management might improve the cultural health of the catchment. Cultural health scores were based on the status of the site, suitability to harvest mahinga kai, physical and legal access, site pressure, degree of modification, and the identification of valued and pest species. These were used with other assessment tools, stream health monitoring, bacterial water quality, and electric fishing. In 2007, when the first assessments were made, the cultural values of the catchment were poor and these did not improve in the 2012 assessment (Lang et al., 2012). Improvement had been expected because of the removal of treated waste effluent from the estuary. Unfortunately, a series of large earthquakes damaged the infrastructure. Raw sewage was released into the rivers and estuary; in addition, sediments were disturbed. These events most likely explain the lack of improvement in cultural health of the habitat.

Cooperative management or co-management

In an early review, Sen and Neilsen (1996) described fisheries co-management as an arrangement where responsibility for resource management is shared between the government and user groups. It is an element of community fisheries management. Governments and scientists have concerns about the impacts of harvesting on populations, and the conflicting perceptions about the status of the shellfish stocks have amplified the need for co-management. Using examples from small-scale fisheries from Africa, Asia, the Caribbean, Europe, North America, and the Pacific, he concluded that most of the examples were at an early stage of implementation and the reason for introducing measures was because of overexploitation of the fishery stock or to resolve conflicts between users. Co-management is seen as one way to increase the resilience in the system to environmental and other changes. This management system depends on the integration of knowledge, the processes used, and the degree of power sharing (Wilson et al., 2006; Berkes, 2009). Three stages are described in co-management (Plummer, 2006; Butler et al., 2012). These are (1) "independence," where there is limited interaction between government and local people; (2) "association," the start of an exchange of information and resource evaluation and shared vision; and (3) "integration," where there is a sharing of the consequences of actions and resolving conflicts. Traditional fisheries often contain unlicensed fishing for general or cultural use, whereas some co-fisheries arrangements are controlled under treaties. In Torres Strait, these include dugong, green turtle (separate management plans for each), and the reef fisheries, which include both fish and invertebrates (Kwan et al., 2006). The objectives of the plans are to achieve sustainability, revive TEK, and allow islanders control in decision making. Although this is an example of co-management, Weiss et al. (2012) suggests that this is dominated by top-down government management. One aspect of co-management is that it depends on resources; for example, in Torres Strait, the fisheries management is well resourced compared with neighboring Melanesian nations (Butler et al., 2012). Successful co-management is a knowledge partnership and can be difficult where it involves indigenous peoples whose knowledge is based on different worldviews (Berkes, 2009). Using science together with traditional knowledge is not simply a synthesis of the two kinds of knowledge, but an ability to develop mutual respect and trust which may not always succeed (Spak, 2005).

Ecosystem-based management

The ecosystem-based management tool was originally developed in response to the impacts of fishing on fish stocks and habitat degeneration due to natural and anthropogenic factors. Fishers were targeted because their communities have biological, oceanographic, economic, social, and cultural aspects which can contribute to the fisheries management (Aswani, 2011). The challenge is to validate the information and create policies and legal ways to integrate information into fishery-management systems (Mathew, 2011). Some suggested ways of doing this included recognizing fishers as holders of knowledge and accepting customary law. It was also suggested that governments and managers need to guarantee the approaches to conservation will be fair and developed under a system of co-management. It was felt that scientists need to overcome their reluctance to use nonspecialist knowledge instead of data. They also need to support the requirements of the communities. Finally, mechanisms need to be established to guarantee that knowledge sharing will benefit and not harm the fishers. Knowledge should be used to conserve the fish stocks and protect the habitat for long-term food security. There should be conflict resolutions in place and the propriety rights of traditional knowledge will remain with the providers.

Integrated coastal management

This is defined as a continuous and dynamic process allowing decisions to be made for the sustainable use of development and protection of coastal and marine areas and resources. The Haida Nation in northern British Columbia has been resource owners and managers for thousands of years on Haida Gwaii, where they apply traditional knowledge and experience to fisheries management. Their approach is based on Haida ethics, principles or values of respect, balance, interconnectedness, seeking wise council, and responsibility. They demonstrate a commitment to responsible and respectful management of marine resources and ecosystems. Plans for the Pacific North Coast Integrated Management Area (PNVIMA) are being led by the Council of the Haida Nation, under the Haida constitution in an example of local community-based and co-governance. The lead agency is Fisheries and Oceans Canada and their approach is for collaborative management with Aboriginal peoples, a change from the previous regime where government policies tended to displace first nations from marine resources (Jones et al., 2010). This initiative was launched in March 2009 and is ongoing, strengthened by court decisions and policies relating to Aboriginal rights and title. This approach has resulted in the establishment of a network of marine-protected areas with levels of protection influenced by first nations and marine resource management that includes principles of social equity and ecosystem justice.

Conclusions

There is overwhelming support for including traditional knowledge in resource management of estuarine and coastal marine areas and an increasing number of examples where this has been shown to benefit all users. Worldwide, the extent of local or traditional knowledge is patchy; some has been lost and in some countries local fisheries knowledge may be largely exclusive to women (Ruddle and Hickey, 2008). There is therefore an urgent need to systematically collect as well as archive cultural, historic, and contemporary information. These different sources of information can then be combined using geo-spatial information systems as suggested by De Freitas et al. (2009). Their multilayer GIS database for artisanal fisheries in Brazil integrates both traditional and scientific data allowing analyses of catch data for target species and highlighting estuarine areas that are likely to be under pressure from overfishing. The GIS tools integrate and translate complex data into an accessible format using maps, and there are widespread applications from such a geodatabase, both culturally and commercially. Also, when updated regularly, they can be used to rapidly respond to changing conditions or emergencies that may require the use of management tools such as temporary closures.

By incorporating traditional knowledge, integrated coastal management is seen as the way forward for the protection and use of marine and estuarine resources (Aswani et al., 2012). The planning and implementing of this management must be backed by legislation that clearly defines the rights of indigenous and government stakeholders. This is the case in New Zealand, where the Treaty of Waitangi is a legal partnership between the British Crown and Māori and the Conservation Act directs the Department of Conservation to establish co-management arrangements with Māori. Mātaitai reserves are authorized by the Minister of fisheries to manage and control seafood harvesting in keeping with local sustainable management practices. Tangata tiaki/kaitiaki recommends bylaws and issues customary food authorizations, and, while commercial fishing may not be allowed, both Māori and non-Māori are allowed to fish in reserve areas. A lack of communication and mistrust can impede negotiations between indigenous and government representatives, and so a framework is required to integrate information in a respectful way (Gratani et al., 2011) and guarantee that the proprietary rights to traditional knowledge remain with the providers. It is hoped that new management systems for seafood will reduce conflicts between users, decrease overexploitation, and revitalize conservation traditions that may have been lost.

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Cross-references

Adaptive Management
Ecosystem-Based Management
Estuarine Habitat Restoration
Fish Assemblages

CYANOBACTERIA

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Synonyms

Blue-green algae

Definition

Cyanobacteria or cyanophyceae (initially known as blue-green microalgae) are nonmotile and planktonic photosynthetic prokaryotes, belonging to the kingdom Eubacteria, division of Cyanophyta. They are common in some extreme environments and occasionally form dense blooms harmful to estuarine environments.

Introduction

Cyanobacteria (photosynthetic prokaryotes) are the Earth's oldest known oxygenic photoautotrophs (Pearl and Otten, 2013). The fossil records of Schopf et al. (2000) suggest that cyanobacteria have been present on earth for at least 3.5 billion years, being distributed worldwide from polar to equatorial latitudes (Vincent, 2000; Wynn-Williams, 2000). Their proliferation during the Precambrian era (~3.5 bya) dramatically altered the previously anoxic biosphere which led to the evolution of higher terrestrial plant and animal life (Schopf et al., 2000). Thus, the cyanobacteria group constitutes a large and morphologically diverse phylum with more than 4,000 isolates and 19 of the most important taxa (species).

Many genera have the ability to fix atmospheric nitrogen (N_2) (through an anaerobic process), while they can store phosphorus (P) and sequester iron (Fe) and a range of essential trace metals (Whitton, 2012). These traits enable them to exploit both nutrient-scarce and nutrient-enriched, diverse terrestrial and aquatic environments worldwide. The cyanobacteria present a range of attributes that give them, in certain environmental conditions, a clear competitive growth advantage over planktonic algae, and therefore they thrive in all kinds of environments (Gomes et al., 2012). Besides planktonic forms, benthic cyanobacteria constitute the principal colonizers at the interface between sediments and water, where they affect fluid flow dynamics and structure formation (Whitton and Potts, 2000). Once classified as microalgae, the cyanobacteria produce photosynthetic pigments (chlorophyll *a* and/or other accessory pigments such as phycocyanin, allophycocyanin, and phycoerythrin) (Briand et al., 2003).

Here, the status of cyanobacteria in estuarine environments is reviewed, their biological and ecological features, and roles in primary sedimentary structures. In addition, the effects of anthropogenic and climate change on cyanobacteria blooms and toxicities are examined.

Ecobiology of cyanobacteria

Cellular morphological features of cyanobacteria are very diverse, including spherical, ovoid, and cylindrical unicellular species, as well as multicellular colonial and filamentous forms (Couté et al., 2001). Some species are able to differentiate specialized cells: (1) heterocysts which are able to fix nitrogen in water under N-limited conditions; and (2) akinetes which tolerate stressful conditions such as periods of high temperature or drought. Cyanobacterial species are sometimes difficult to identify due to their high phenotypic plasticity (Briand et al., 2003).

These organisms comprise a unique phylogenetic group of bacteria that perform oxygenic photosynthesis (Hackenberg et al., 2011). In addition, cyanobacteria occupy diverse ecological niches and exhibit enormous diversity in terms of their habitats, physiology, morphology, and metabolic capabilities (Beck et al., 2012). In fact, cyanobacteria are able to establish competitive growth in almost any environment where there is, at least temporarily, water and sunlight (Badger et al., 2006; Esteves-Ferreira et al., 2013).

The most recent taxonomic classification of cyanobacteria is based on the so-called polyphasic approach (Johansen and Casamatta, 2005). In this approach, molecular phylogenetic analyses are the basic criteria for classification of genera and species, with the cytological and morphological markers (synapomorphic and autapomorphic characters) and the ecology (habitat preference, life strategy, and ecophysiology) considered an integral part of the taxonomic definition, with additional important biochemical and molecular markers (Komarek and Mares, 2012).

Two morphological types are distinguished within the cyanobacteria group: (1) the filamentous species forming elongated cell chains (trichomes) often bundled together (multi-trichomous species); and (2) the coccoid species forming spheroidal cells often arranged in cell clusters (Staley et al., 1989; Whitton and Potts, 2000). In estuarine ecosystems, cyanobacteria are primary producers that use light energy to synthesize organic matter from mineral nutrients and CO_2 (photosynthesis). Their specific physiologic capabilities enable them to compete very efficiently with other photosynthetic microorganisms and to regulate their buoyancy (by means of gas vacuoles). Thus, they can colonize different depths in the water column depending on the location of nutrients and availability of light (Klemer et al., 1982; Walsby et al., 2001).

Their possession of accessory pigments, such as phycoerythrin, enable several cyanobacteria species to carry out photosynthesis at depths that receive only green light and where, in addition, nutrients are more abundant than at the surface (as in the case of surface waters rapidly depleted following spring algal proliferations). Cyanobacterial pigments, as well as mycosporin-like amino acids, are involved in their capacity to resist ultraviolet radiation in surface waters, giving them another advantage over some phytoplankton (Castenholz et al., 2000).

The cyanobacteria are poorly grazed by zooplankton due to their production of mucilage layers (Mur et al., 1999; Goleski et al., 2010). Recent data reveal that cyanobacteria have adopted a mode of defense depending on grazer pressure (i.e., they are able to modify their defense reaction according to the actual risk of grazing) (Gomes et al., 2012). The synthesis of different toxins by many cyanobacterial blooms gives them a selective advantage, since some zooplanktonic predators are susceptible to these toxins and thus avoid eating cyanobacteria (Jacquet et al., 2004; Oliver et al., 2010).

The biology and ecology of cyanobacteria have been extensively studied throughout the world during the two last decades due to their expansion and proliferation in most aquatic environments (Pearl and Otten, 2013). Several inner ecophysiological strategies allow the cyanobacteria to exploit anthropogenic modifications of aquatic environments (specifically nutrient over-enrichment and hydrologic alterations). Thus when conditions of light and water column stability are favorable, cyanobacteria may proliferate creating a competitive advantage over other species of phytoplankton (Smith and Bennet, 1999; Peter et al., 2002). Because benthic cyanobacteria are the principal colonizers of the interface between sediments and water, they can affect fluid flow dynamics and structure formation in biofilms and microbial mats. Therefore, they greatly influence the sedimentary dynamics of peritidal depositional systems as noted by Vincent et al. (2000) and Noffke et al. (2003).

Cyanobacteria status in estuarine ecosystems

Due to their salinity gradients, estuaries provide a large variety of aquatic habitats for native populations of marine, brackish, and freshwater planktonic species (Telesh, 2004). The spatial zoning and functional characteristics of estuaries result in biologically active zones with high concentrations of bacteria and microalgae (Golubkov et al., 2001). Further, the microbial communities of estuarine ecosystems are susceptible to rapid changes in response to the flux of environmental conditions. Thus, the flux of dissolved and suspended organic and inorganic material, in addition to hydrological variations, significantly affects microbial abundance, diversity, and activity in the estuarine ecosystems (Bouvy et al., 2010).

Given sufficient nitrogen inputs, estuarine and coastal marine environments can be driven by phosphorus limitation which contributes to greater far field nitrogen enrichment and eutrophication at greater distances (Howarth et al., 2011). Nutrient loading from coastal watersheds and upstream systems typically deliver higher quantities of nutrients than those entering from coastal ocean waters (Galloway et al., 2004; Fennel et al., 2006).

In estuarine waters with salinities greater than 8–10 ppt, planktonic cyanobacteria capable of N-fixation are largely absent (Howarth and Marino, 2006; Marino et al., 2006; Howarth and Paerl, 2008). A decrease in planktonic N-fixation in estuaries has been attributed in part to high levels of sulfates in seawater, making the assimilation of molybdenum (an element required for N-fixation) difficult. This leads to slow potential growth rates of N-fixing cyanobacteria (heterocystus cyanobacteria, where N-fixation occurs only in heterocyst cells) exposed to grazing by zooplankton and benthic animals (Chan et al., 2006).

Cyanotoxicity and cyanobacterial blooms

A notable increase in occurrence and intensity of cyanobacteria toxic blooms has been observed worldwide over the last several decades (Eiler and Bertilsson, 2004; Pearson and Neilan, 2008; Rinta-Kanto et al., 2009). For major cyanobacterial genera involved in harmful blooms, the optimal growth rates and bloom potentials have increased with higher water temperatures; thus global warming may be playing a key role in the expansion and persistence of bloom-forming cyanobacterial taxa (Pearl and Fulton, 2006).

A recent study by Pearl et al. (2013) showed how cyanobacterial surface blooms may locally increase surface water temperatures due to light energy absorption via an array of photosynthetic and photoprotective pigments (chlorophylls, carotenoids, and phycobilins). This represents a positive feedback mechanism by which cyanobacterial bloom species can optimize their growth rates leading to competitive dominance over eukaryotic phytoplankton. Global warming, therefore, may enhance cyanobacterial dominance in the plankton as reported by Bonilla (2012).

Cyanobacterial blooms are complex microbial assemblages, consisting of many representatives from characterized phyla (Pope and Patel, 2008; Li et al., 2011; Wilhelm et al., 2011). The morphological features of organisms within a bloom appear as associative microbial assemblages analogous to biofilms (Zehr et al., 1995; Reid et al., 2000; Omoregie et al., 2004; Burke et al., 2011).

The initiation, maintenance, and subsequent decline of cyanobacteria blooms depend to a large extent on the availability of nitrogen (N) and phosphorus (P) (Levich, 1996). It also depends on the ratios of N and P, selecting for organisms capable of fixing atmospheric nitrogen over those lacking this physiology (Klausmeier et al., 2004).

Among the harmful cyanobacteria species cited elsewhere, the most common toxin producing cyanobacteria N_2-fixing genera are *Anabaena*, *Aphanizomenon*, *Cylindrospermopsis*, *Lyngbya*, *Nodularia*, *Oscillatoria*, and *Trichodesmium*, while the non-N_2 fixers are *Microcystis* and *Planktothrix* which thrive in fresh and estuarine environments as well as in marine systems (Pearl et al., 2013). The major harmful toxins produced by toxic or harmful cyanobacteria are large classes of natural polyketides compounds, nonribosomal peptides, or a mixture of both (Moreira et al., 2013). Their biosynthesis is performed by a family of multi-enzymatic complexes called nonribosomal peptide synthetases (NRPS) and polyketide synthases (PKS) organized into repeated functional units known as modules (Carmichael, 1992; Cane et al., 1999).

The "harmful" environmental aspect of cyanobacterial blooms described by Pearl et al. (2013) begins with a loss of water clarity, suppression of aquatic macrophytes, and negative effects on invertebrate and fish habitats. Consequently, the bacterial decomposition of dying blooms may lead to oxygen depletion (hypoxia and anoxia) and subsequent fish kills.

Smith et al. (2008) also indicates that cyanobacterial odorous and bioactive metabolites have a negative impact on aquaculture organisms. The toxins cause mortality of aquaculture organisms or harm consumers consuming the seafood products via accumulation of hepatoxins, cytotoxins, neurotoxins, dermatoxins, and brine shrimp/ molluscan toxins. Some metabolites degrade the nutritional state of aquaculture species (inhibitors of proteases and grazer deterrents). Aquaculture species or aquaculture workers can be seriously impacted by dermatoxins, irritant toxins, hepatoxins, and cytotoxins.

The cyanobacterium *Microcystis aeruginosa* is the most common bloom-forming and hepatotoxin-producing species of cyanobacteria. It is known to produce the hepatotoxic heptapeptide microcystin in a variety of forms (Kaebernick et al., 2000). Microcystin binds to the multispecific bile acid transport system, subsequently causing toxic effects on hepatocytes. The effect is the inhibition of eukaryotic protein phosphatases PP2A and PP1 (Ppp1, Ppp2, Ppp4, Ppp5, and Ppp6) that are involved in tumor promotion and genotoxicity (Moreira et al., 2013).

The most recent data on microcystin is that it occurs worldwide. Anthropogenic nutrient loading, rising temperatures, enhanced vertical stratification, and an increase in residence time favor cyanobacterial dominance and CyanoHAB proliferation in a wide range of aquatic ecosystems (Pearl and Otten, 2013).

Summary

Cyanobacteria can adapt to dramatic changes in hydrobiological conditions. They have numerous physiological adaptations and mechanisms that enable them to take advantage of environmental changes and extremes that influence the biosynthesis of cyanotoxins for several cyanobacterial species.

The occurrence of harmful cyanobacteria has been linked to an increase in nutrient pollution in aquatic ecosystems. Future climate change is predicted to cause shifts in species composition of cyanobacterial blooms favoring invasive species since modern global distributions of cyanobacterial species result from differences in evolutionary adaptations and phenotypic traits.

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Cross-references

Algal Blooms
Phytoplankton
Phytoplankton Blooms
Toxic Blooms

D

DELTA PLAIN

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Definition

A delta plain is a low-elevation floodplain formed at the mouth of a river.

Characteristics

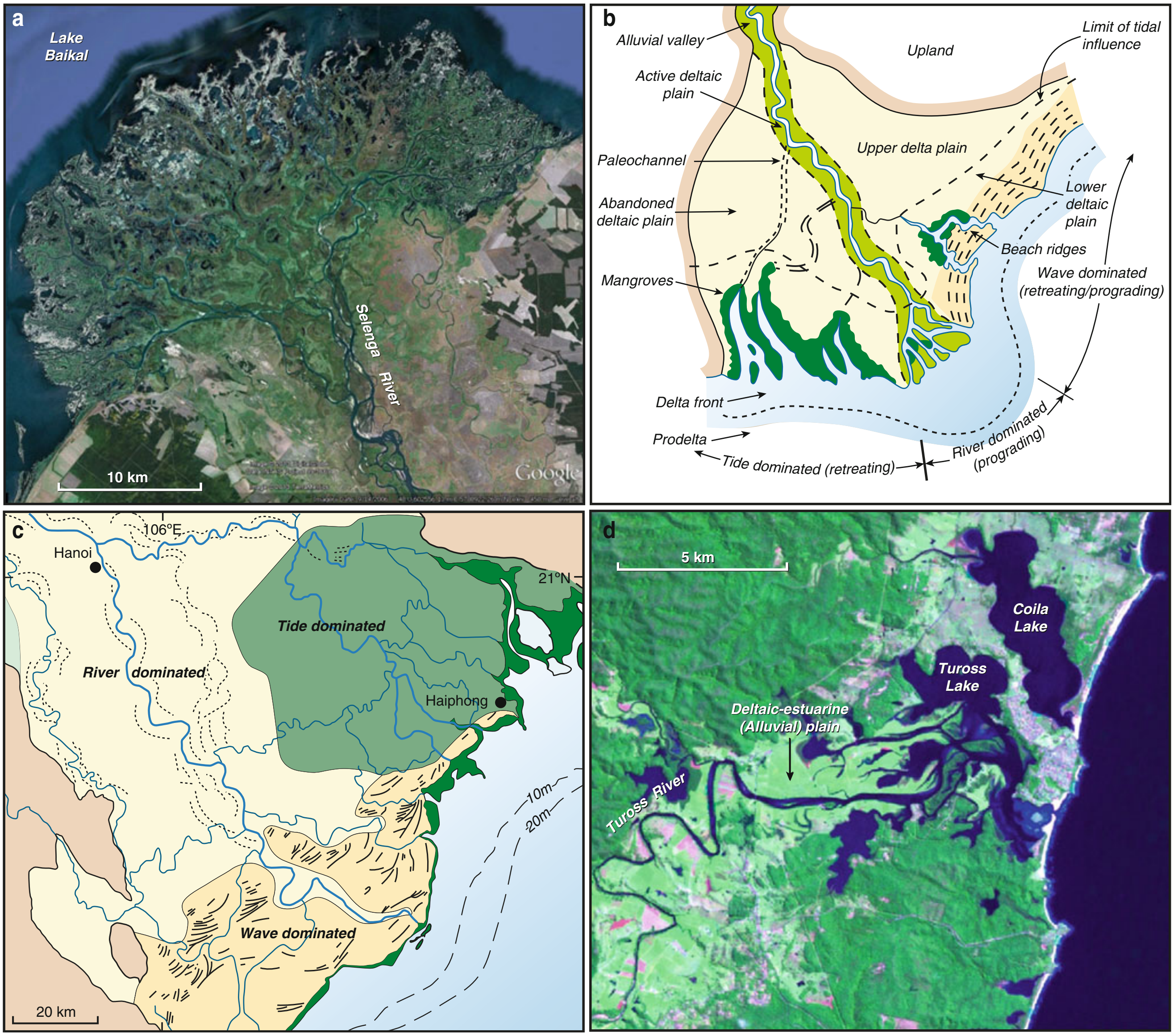
A delta plain is one type of low-lying coastal plain, formed where a river empties into the sea (or, rarely, into a freshwater body, as in the case of the Selenga Delta, Figure 1a). Large deltas can generally be subdivided into an upper deltaic plain influenced primarily by fluvial processes and a lower deltaic plain, dominated by wave and tidal processes (Figure 1b). The river flows through the "active" section, but there is commonly an abandoned section containing paleochannels marking former river courses (Wright et al., 1974).

The Red River Delta in northern Vietnam (Figure 1c) can be differentiated into a river-dominated upper delta plain where channels are flanked by levées and meander scroll bars marking former river courses, a southern wave-dominated section with sequences of shore-parallel beach ridges, and an eastern tide-dominated section with numerous tapering tidal creeks (Mathers and Zalasiewicz, 1999).

Similar near-horizontal alluvial plains can form along estuaries, sometimes called deltaic-estuarine plains. For example, coastal lagoons (e.g., Coila Lake, Figure 1d) and barrier estuaries (e.g., Tuross Lake) become gradually infilled as fluvial sediment builds a bayhead delta into the estuarine basin.

Extensive, perennially or seasonally flooded, wetlands may characterize delta plains in their natural state (Figure 1a). Megadeltas in southeastern Asia are often the location for intensive rice cultivation, but also support megacities (e.g., Hanoi, Figure 1c), many of which require augmentation of levées for flood mitigation. The ease with which land can be cleared and the fertility of soils has encouraged their agricultural use (e.g., Figure 1d).

M.J. Kennish (ed.), *Encyclopedia of Estuaries*, DOI 10.1007/978-94-017-8801-4,



Delta Plain, Figure 1 (**a**) The Selenga River, draining into Lake Baikal, has bifurcated into numerous distributaries that dissect delta plain wetlands (Source Google Earth, © DigitalGlobe); (**b**) the principal components of a large delta plain, for example, (**c**) the Red River Delta plain (After Woodroffe and Saito, 2011); (**d**) the extensive plains flanking the Tuross River, as it drains into a barrier estuary in southeastern Australia, are much better developed than those where a smaller creek empties into Coila Lake, a coastal lagoon (Image: © Commonwealth of Australia, ACRES, Geoscience Australia).

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Cross-references

Deltas

DELTAS

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Definition

A delta is a discrete shoreline sedimentary protuberance formed where a river enters an ocean, a semi-enclosed sea, an estuary, a lake, or lagoon and supplies sediment more rapidly than it can be redistributed by basinal processes (modified after Elliott, 1986).

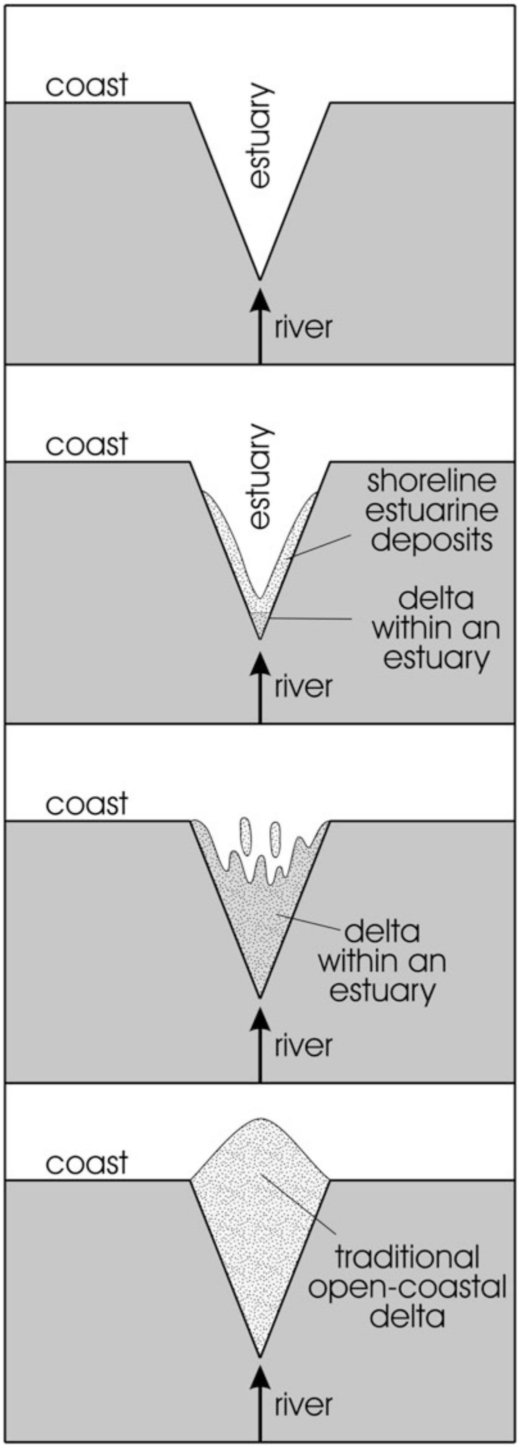
Deltas in an estuarine environment

A delta is often closely associated in time and space with an estuary (Figure 1), but frequently in the literature the two are not adequately separated, particularly for tide-dominated estuaries. For the same riverine outlet, a delta is a geomorphic and sedimentologic feature, while an estuary is a hydrochemical one where riverine freshwater flowing into a bay, a lagoon, or semi-enclosed coastal body of water mixes with seawater (Cameron and Pritchard, 1963; Pritchard, 1967; Day, 1981). Deltas may have either a perennial or a seasonal freshwater flow and hence a perennial or seasonal freshwater-to-seawater transition resulting in some parts of them being estuarine.

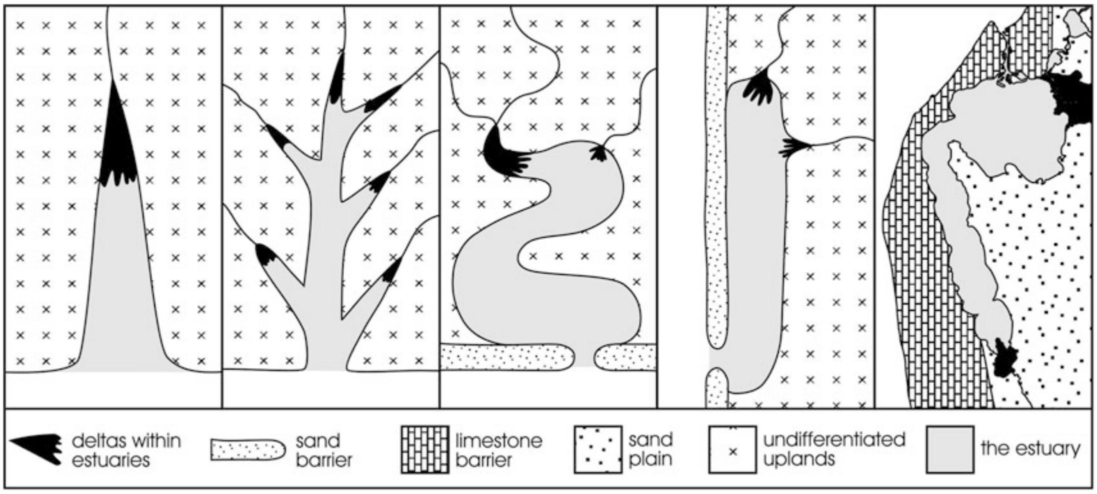
To a large extent, all deltas can be estuarine in the sense that some part of them will have a freshwater-to-seawater transition, and large estuarine environments whose basin has not been filled with sediment may contain small-scale deltas along their margins or in their headwaters (Figure 2). Geomorphologists and sedimentologists, focused on landforms and stratigraphy, generally do not deal with the hydrochemical estuarine components of deltas, and conversely, researchers of estuarine ecology, hydrochemistry, or hydrodynamics generally have focused on deltas in an estuary only in terms of geomorphology, sedimentology, or stratigraphy. This difference of emphasis becomes important here because the deltas described in this contribution are those occurring in the context of a larger estuarine setting: as such, a "delta within an estuary" is distinguished from an "estuary within a delta" (Figure 3). This contribution focuses on the "delta within an estuary."

Deltas within estuaries generally are relatively small sedimentary accumulations compared to the size of their estuarine setting (Figure 2). They have been variably termed as "bayhead deltas" (*cf.* van Heerden and Roberts, 1988; Dalrymple et al., 1992; Kindinger et al., 1994), "river deltas" (Hayes, 1975), and "intra-estuarine deltas" (Semeniuk et al., 2011). As not all of them are located in "bayheads," the term "intra-estuarine delta" is used here for those deltas occurring within estuaries.

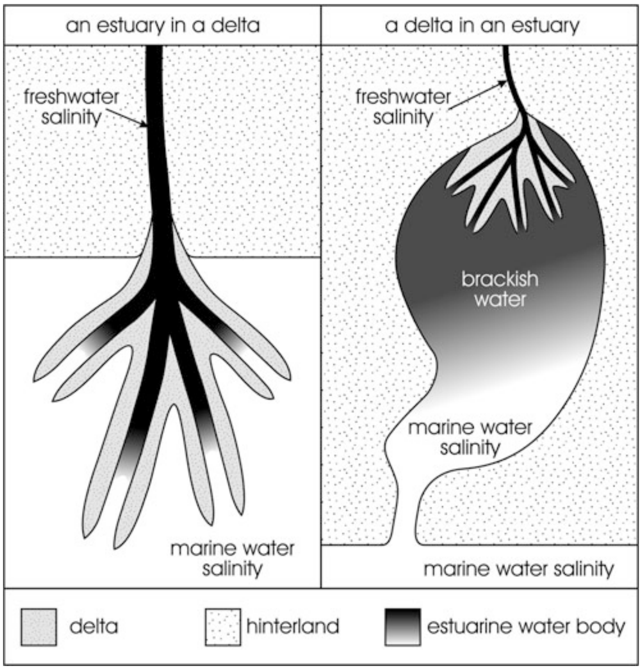
In contrast, deltas in open coastal settings generally are large sedimentary accumulations but are relevant to smaller deltas that occur within estuaries in that the principles involving hydrodynamics, geometry/morphology, mechanisms of construction, sedimentology and facies, and



Deltas, Figure 1 Idealized diagram showing the gradation from a relatively narrow v-shaped open estuary with minimal or no sedimentary fill to sediment-filling estuaries to a coastal delta where sedimentary accretion has prograded into the marine environment. Intra-estuarine deltas are present and more clearly evident in estuaries where sediments have not fully occluded them.



Deltas, Figure 2 Idealized diagram showing a range of estuary types, from an incised single valley to rias, a flooded valley on a coastal plain, a barred estuarine coastal lagoon and a compound estuary, and the occurrence of intra-estuarine deltas (*black*) therein.



Deltas, Figure 3 Idealized diagram showing the dual concepts of an estuary within a large delta and a delta within a large estuary (or an intra-estuarine delta). In each example, the field of salinity is freshwater = *black*, brackish water = *gray*, and marine water = *white*.

stratigraphy are similar. The Mississippi Delta complex, Nile Delta, Niger Delta, São Francisco Delta, Klang Delta, and Fly Delta are examples of large open coastal deltas (Allen, 1970; Coleman et al., 1970; Gould, 1970; Summerhayes et al., 1978; Dominguez, 1996; Baker et al., 2009). Such deltas have been classified as to their plan geometry in response to their hydrodynamic setting as fluvial-dominated deltas, tide-dominated deltas, and wave-dominated deltas (Galloway, 1975) or by their depositional architecture and facies (Postma, 1990). For completeness in the descriptions of deltas, the reader is referred to geomorphic and stratigraphic descriptions of such open coastal deltas in Scruton

(1960), Morgan (1970), Wright and Coleman (1973), Galloway (1975), Coleman (1976), Reineck and Singh (1980), Elliott (1986), Nemec (1990), Postma (1990), and Hori and Saito (2003).

Factors determining types of deltas in estuaries

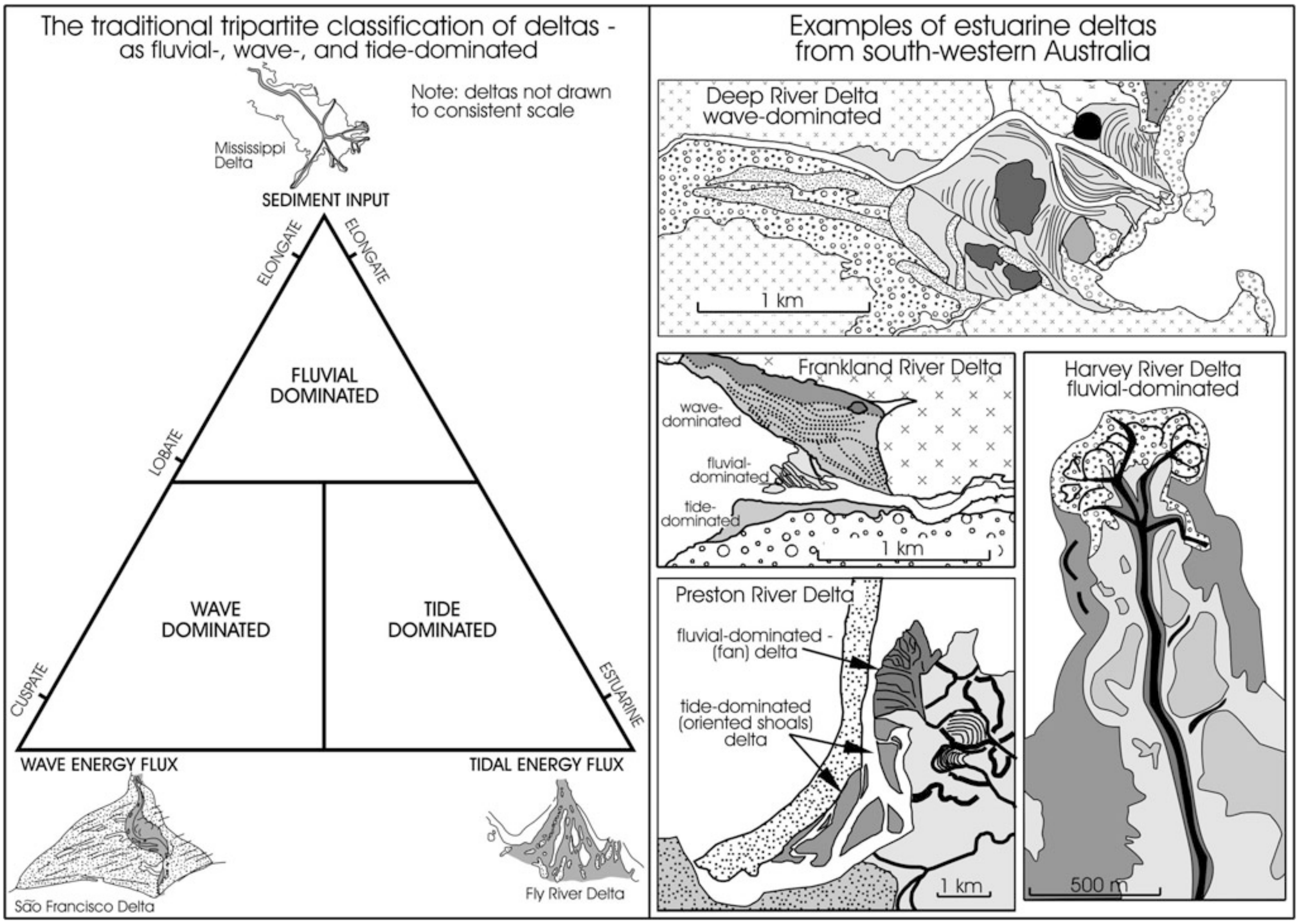
Depending on the size and shape of the estuary, a delta within an estuary can be variable in terms of plan geometry (morphology), landforms within the delta, sedimentary facies, and stratigraphy. The main factors determining the morphology and landforms of deltas in estuaries are (1) seasonality and strength of river flow, (2) the salinity of the receiving estuarine basin, (3) the magnitude of tides, (4) wind and wind waves, and (5) the shape of the estuary and where the river(s) is/are located. A number of these factors are interrelated and combine to produce a given delta type or delta form. For instance, the seasonality and strength of river flow can affect the salinity of the receiving basin in that strong perennial river flow will ensure that the receiving basin is perennially brackish, particularly where the tidal regime is microtidal. Similarly, the extent to which wind and wind waves can influence delta morphology and landforms can be dependent on the external shape of the estuary and the location of the river(s) in relation to the wind field and wave field.

The seasonality and strength of the river flow determines whether the delta will be fluvial dominated, tide dominated, or wave dominated. Perennial rivers, deriving from large drainage basins in humid climates, with strong river flow (and, commonly, concomitant strong sediment transport), produce fluvial-dominated conditions at the river outlet. In this setting, delta morphology, controlled by fluvial conditions, tends generally to be fan-shaped varying to elongate and digitate. With fluvial-dominated conditions, the salinity of the estuarine-receiving basin also can play an important part in determining the style and course of river flow into the estuary and hence the shape of any deltaic sedimentary accumulation. In this context, it should also be noted that the dynamics of sediment-laden river flow with its various amounts of traction load and/or suspension load entering and interacting with an estuarine-receiving water body of different density (ranging from fresh to brackish to marine) will result in different types of deltaic depositional morphology. Bates (1953), Wright (1978), and Orton and Reading (1993) describe this variability of depositional styles and resulting delta forms in relationship to three situations: (1) hypopycnal flows in which density of the suspended sediment flow is less than that of the receiving estuarine water body, (2) homopycnal flows in which density of the suspended sediment flow is equal to that of the receiving estuarine water body, and (3) hyperpycnal flows in which density of the suspended sediment flow is more than that of the receiving estuarine water body. Hypopycnal, homopycnal, or hyperpycnal flows also determine the nature of river mouth dynamics as to whether buoyant, inertial, or frictional factors are dominant in distributing and shaping the sediment plume and sand bars (Bates, 1953; Coleman, 1976; Wright, 1978) and the shape of any freshwater jet as it enters a more saline estuary (Wright, 1978).

Riverine freshwater flowing into an estuarine basin of denser brackish water or marine salinity will exhibit hypopycnal flow, with freshwater overlying the denser estuarine water. Riverine freshwater flowing into an estuarine basin of similar freshwater, or turbid freshwater flowing into brackish water, will exhibit homopycnal flow, with the river water invading the estuarine water of similar density in a turbulent mixing front. At the other extreme, sediment-laden turbid riverine freshwater flowing into an estuarine basin of freshwater or weakly brackish salinity will exhibit hyperpycnal flow, with the denser sediment-laden river flow (comprising sediment in suspension and transported in traction) forming a base flow under the less-dense estuarine water.

However, where riverine input is seasonal, or where the sediment-transporting river flow is inter-annual, tides and wind waves will predominate as the formative agents in delta type and in the development of its plan geometry. The flux of tides on a daily or semidiurnal basis can have a prevailing influence on determining delta shape, and river sediment delivered to the mouth of the river in a mesotidal or macrotidal estuary will be redistributed and sculptured by tidal currents and shaped into tidal-current-elongated shoals. In regions with strong winds, wind waves are generated on estuarine water bodies and impinge on delta fronts. Depending on fetch, and particularly if the river mouth is downwind in an estuary with a large fetch, deltaic sediments deposited at the river mouth will be subject to prevailing wind waves. As a result, wave-dominated deltas will develop.

The shape of the estuary and position of the river (s) within it play important roles in determining delta morphology. Relatively simple estuaries, that are v-shaped, narrow linear valley tracts with a single river mouth at the head of the estuary, are subject to interactions of river flow, tidal flux, and wind waves, and the delta developed at the estuary head will tend to be the form indicative of the locally dominant hydrodynamic condition. Complex estuaries and large estuaries with large fetch, on the other hand, can create conditions where there are complicated hydrodynamics of waves and wind-induced currents, and in situations where there is more than one river entering the estuary, each river may be subject to differing hydrodynamics. There will also be differences in the deltas where the rivers have dissimilar flow magnitudes. Within the one estuarine basin with multiple river inflows and multiple deltas, one delta may be fluvial dominated; others may be tide dominated or wave dominated. The deltas in these types of complex estuaries are even more variable if the various contributing rivers arise from different geological provenances in their respective hinterland and are delivering different suites of sediments.



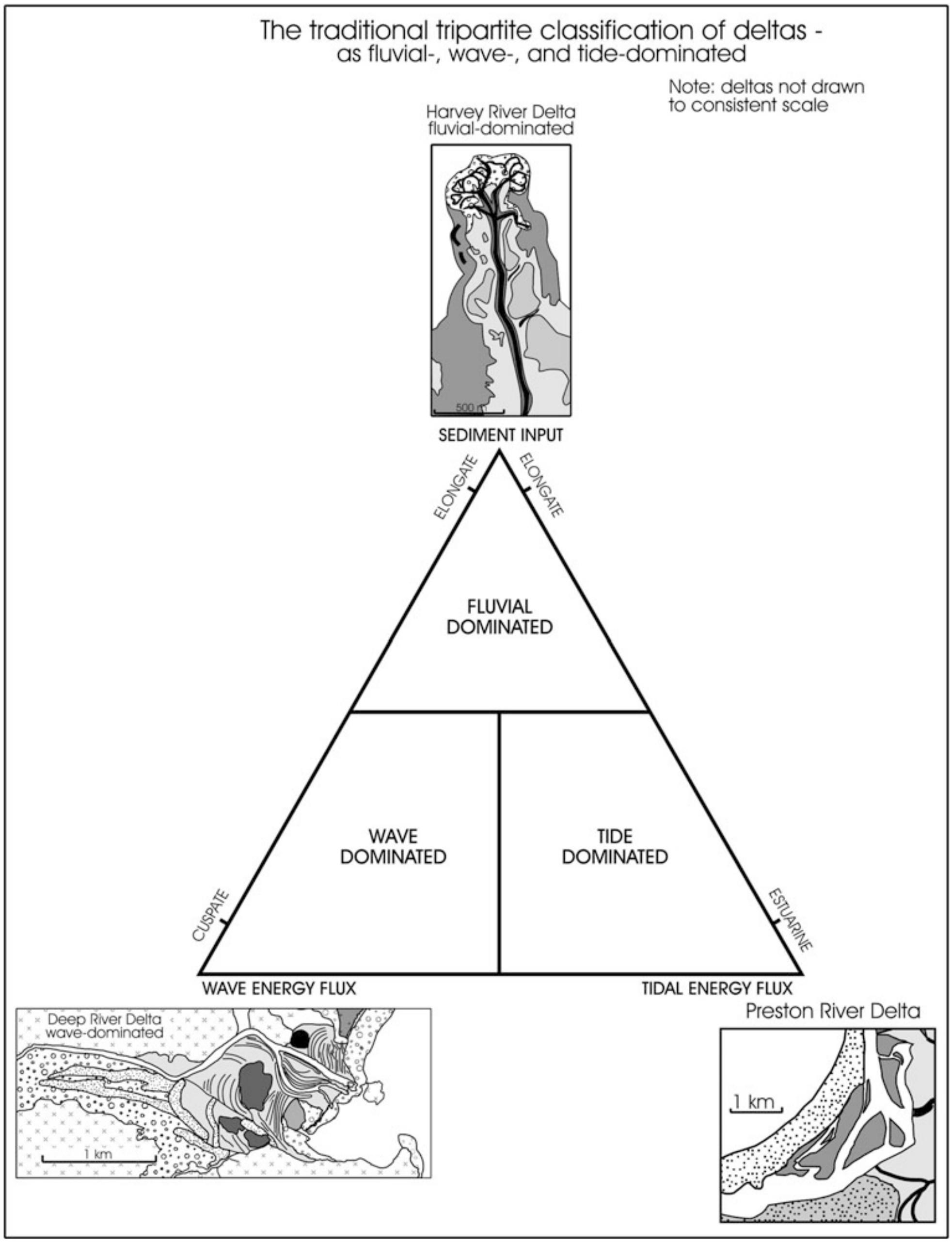
Deltas, Figure 4 The traditional tripartite classification of deltas (Modified from Galloway, 1975; Reineck and Singh, 1980) based on open marine deltas and showing the Mississippi Delta, the São Francisco Delta, and the Fly Delta as examples of fluvial-dominated, wave-dominated, and tide-dominated deltas, respectively. Examples also are shown of intra-estuarine deltas from southwestern Australia to illustrate fluvial-dominated forms (Harvey River Delta), wave-dominated forms (Deep River Delta), partly tide-dominated and fluvial-dominated forms (Preston River Delta) and a delta that shows longitudinal sectors that are fluvial dominated, wave dominated, and tide dominated (Frankland River Delta).

Types of deltas within estuaries

Given that a delta is a sedimentary deposit formed where a river enters an ocean, a semi-enclosed sea, an estuary, a lake or lagoon, the hydrodynamic forces operating on these sediments to distribute, rework, and shape them into various types of deltaic bodies are river flow (fluvial hydrodynamics), waves, and tides. If any of these hydrodynamic forces are dominant, the resulting delta will be fluvial dominated, wave dominated, or tide dominated (Figures 4 and 5). A classification of deltas based on their resultant morphology deriving from the style of hydrodynamic forcing was developed by Morgan (1970), Wright and Coleman (1973), Galloway (1975), and Coleman (1976). While the emphasis on delta classification in the literature has been on open coastal deltas, the classification can equally be applied to deltas wholly contained or confined in semi-enclosed water bodies, estuaries, lakes, and lagoons.

Deltas can be subdivided into various geomorphic/sedimentologic units related to location within the delta. These geomorphic/sedimentologic units can vary in size from delta to delta, and not all may be present in every delta. From the river hinterland to the deeper water into which the delta progrades, the units are (Coleman and Wright, 1975; Hart, 1996) alluvial feeder, upper delta plain, lower delta plain, delta front, delta slope, and prodelta. The mechanics of delta formation in different hydrodynamic situations are well summarized by Hart (1996).

Where fluvial processes dominate over the two other hydrodynamic forces, the resulting delta is termed a “fluvial-dominated delta.” Its morphology is determined by river flow transporting sediment loads in traction and suspension, and, depending on the salinity of the receiving estuarine water body and the nature of influx (whether hypopycnal, homopycnal, or



Deltas, Figure 5 The traditional tripartite classification of deltas of Figure 4 with some intra-estuarine deltas from southwestern Australia on the ternary diagram.

hyperpycnal), the delta can be lobate, fan-shaped, elongate, or digitate. The delta progrades into the estuary by subaqueous deposition of a fan of sand or bar-finger sand, shoaling to high-tidal levels and river flood levels. The fans of sand or bar-finger sands are capped by tidal deposits, levee deposits, and floodplain deposits, while interdistributary bays are filled with tidal flat deposits shoaling to floodplain deposits. The overall delta form consists of (1) a subaerial part whose plan shape is lobate, fan-shaped, elongate, or digitate and whose geomorphic/sedimentologic components include levee banks, floodplains, high-tidal to supratidal flats, lagoons, and abandoned channels; (2) a delta slope comprised of sheets, fans or bar-fingers of sand or muddy sand, and laterally intervening bays underlain by sand or mud; and (3) a prodelta usually underlain by mud that forms a peripheral apron around the delta slope.

Where wave action is dominant because the delta resides in an estuary with a strong component of wind and wind-generated waves, regardless of the mechanism that delivers sediment to the front of the river mouth (*viz.*, hypopycnal, homopycnal, or hyperpycnal), the sediment deposited at the river mouth is subsequently reworked shoreward into a series of beach ridges, or recurved spits, or bars and their leeward lagoons, all built by waves and wind to levels of the high tide and above. Progradation of the delta thus is by beach ridge accretion, recurved spit accretion, or as a series of bars and lagoons. The delta usually is a lobate complex of prograded beach ridges, a series of beach ridges and/or recurved spits with intervening swales and/or linear lagoons, or a prograded series of bars and linear, oval to circular lagoons. The beach ridges, recurved spits, and bars are underlain by sand and are often accreted to above the level of high tide. The swales and linear, oval to circular lagoons, depending on the style of sedimentary filling, are underlain by sand, muddy sand, mud, or peat. River floods, unable to reach the height of beach ridges, are confined to the distributary channels or flood into the beach ridge swales.

Where tides are the dominant hydrodynamic force, again, regardless of the mechanism that delivers sediment to the front of the river mouth, the sediment is subsequently reworked by tidal currents into tidal-current-aligned (usually shore-normal) subaqueous sand shoals that accrete vertically to levels of the high tide. Progradation of the delta thus is by tidal shoal vertical accretion to a level where the deposits are finally capped by floodplain sediments. The delta front (or delta slope) usually is a crenulate to palmate complex of prograded subaqueous to tidal shoals, and the landward part of the delta is a floodplain.

Locally in estuaries, where river gradients are relatively steep, there may be development of Gilbert-type deltas. These are a specific type of fluvial-dominated delta, usually fan-shaped and coarse-grained, with internal geometry of simple large cross-stratification corresponding to the delta morphology of topset, foreset, and bottomset (Postma, 1990).

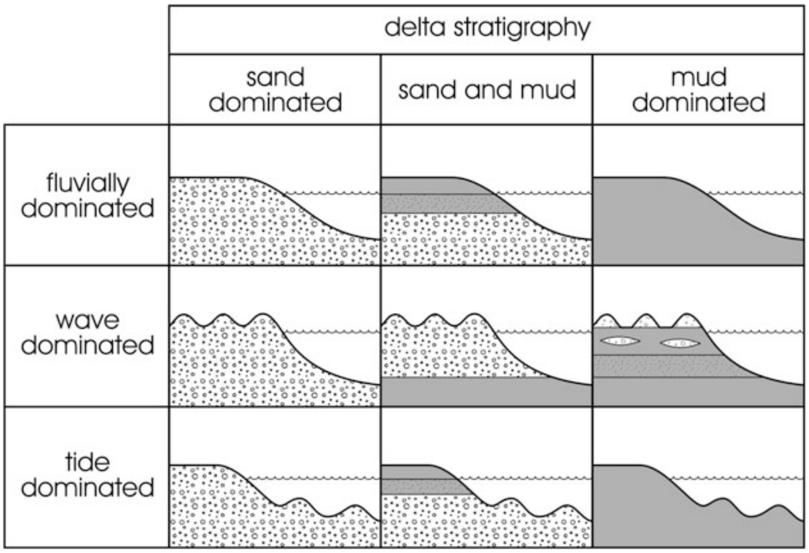
While deltas can be classified as fluvial dominated, tide dominated, or wave dominated depending on their hydrodynamic setting, often in estuaries, because of the complexity of the hydrodynamics, an intra-estuarine delta may exhibit different morphology in different parts of the delta or contrasting landforms reflective of hydrodynamic conditions in a specific part of that delta. For instance, the wave-dominated intra-estuarine delta of the Deep River in southern Western Australia (the Walpole-Nornalup Inlet Estuary; Semeniuk et al., 2011) comprises two distinct geomorphic responses reflecting different degrees of wave action and sediment transport/mobility. There are prograded beach ridges on one half of the delta and a prograded bar-and-lagoon complex on the more sheltered other half. In the same estuary, another intra-estuarine delta (the Frankland River) also reflects the variable hydrodynamic forces across the delta depositional environment. It comprises a wave-dominated part in its northern third (prograded beach ridges and inter-ridge swales), a fluvial-dominated part in its central third (prograded and shoaled digitate/palmate sedimentary accumulation), and a tide-dominated part in its southern third (prograded sand platform). Within an intra-estuarine delta in another estuary in southwestern Australia (the Leschenault Inlet Estuary; Semeniuk, 2000), one side of a fluvial-dominated palmate delta faces a 12 km fetch, and during intermittent winter storms deriving from the north-west, waves break on the shore to create a repetition of storm-wave-generated cheniers across the floodplain.

In hydrodynamically and geomorphically complex estuaries with multiple river entries, there may be a range of intra-estuarine types within the same estuary. For instance, the Peel-Harvey Estuary of southwestern Australia, with three river entries (Semeniuk and Semeniuk, 1990a; Semeniuk and Semeniuk, 1990b), has two wave-dominated deltas (composed of prograded bar-and-lagoon complexes) because they face the prevailing regional summer breezes that generate wind waves on the estuarine water body and one fluvial-dominated delta (composed of prograded fans of sand, levee deposits, and floodplain deposits) that is not subject to these wind waves. The Leschenault Inlet Estuary, with two river entries (Semeniuk, 2000), has one fluvial-dominated delta (a palmate delta) and another delta that is, in part, tide dominated (composed of tidally-aligned shoals) and, in part, fluvial dominated. The Walpole-Nornalup Inlet Estuary of southern Australia, with three river entries (Semeniuk et al., 2011), has two wave-dominated estuaries (composed of prograded bar-and-lagoon complexes, or of beach ridges) because they face the prevailing regional summer breezes that generate wind waves on the estuarine water body and one hydrodynamically complex delta that is one third wave dominated (facing the wind waves generated by sea breezes), one third fluvial dominated, and one third tide dominated (the latter two not subject to wind waves).

Factors determining the stratigraphy of deltas in estuaries

Deltas within estuaries exhibit a variety of stratigraphic sequences, depending on the sediments available, and their hydrodynamic setting. However, given that estuaries as enclosed to semi-enclosed coastal bodies of water where multiple interactions between marine and riverine environments may take place, there are other factors that result in a richness and variety of stratigraphy in intra-estuarine deltas. These differences not only occur between deltas from different estuaries but even between deltas within the same estuary. The main factors determining the stratigraphy within a delta are (1) the provenance of the contributing rivers and types of sediment entering the estuary, (2) seasonality and strength of river flow, (3) nature of tides, (4) contribution of wind and wind waves, and (5) climate.

The provenance of the contributing rivers entering the estuary, that is, the geology of the drainage basins,



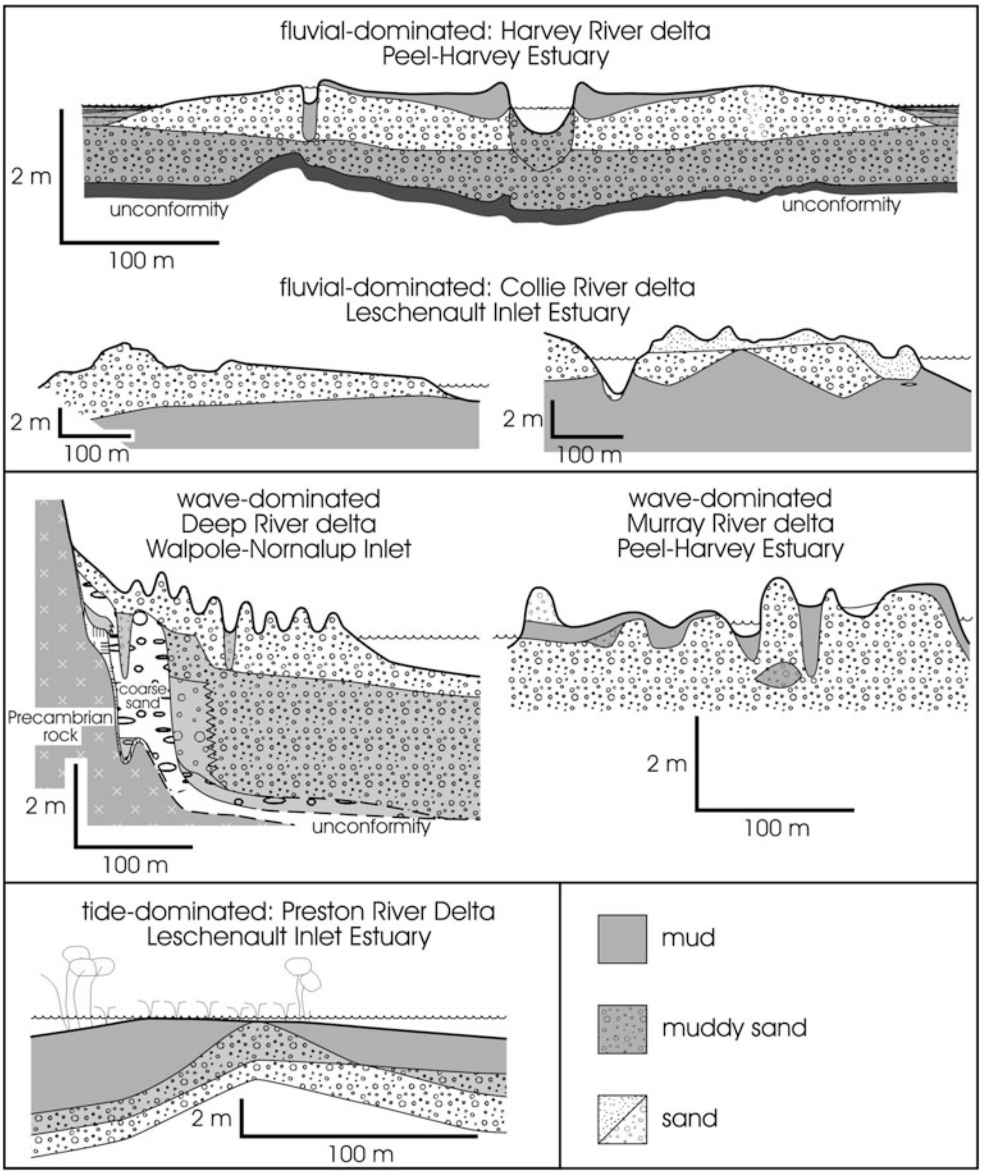
Deltas, Figure 6 Simplified and idealized stratigraphy of deltas formed in sand-dominated, mixed sand-and-mud, and mud-dominated settings under hydrodynamics conditions of fluvial dominated, wave dominated, or tide dominated. The lithologies are simplified to sand, muddy sand, and mud.

determines the supply and composition of sediment delivery. Whether the delta is sand, mixed sand and mud, or mud-dominated, how much gravel is present and the type of shoaling stratigraphy that is developed are factors dependent not only on provenance but also on the hydrodynamic factors at the interface of river and estuary. The effects of seasonality and strength of river flow, the magnitude of the tides, and the influence of wind in generating wind waves and circulation currents on intra-estuarine deltaic stratigraphy are expressed in the dominant grain size which results in, by the way in which the gravel, sand, and mud are separated into different deltaic environments, the distance the gravel, sand, and mud are transported from the river mouth, the types of sedimentological mechanisms that occur, and the range of sedimentary structures that are generated. Differences can result in upward shoaling, sediment interlayering, the development of environment-specific sedimentary structures, and the micro-stratigraphic and macro-stratigraphic sequences.

Climate plays a part in the development of deltaic stratigraphic sequences in that rainfall and evaporation can determine the nature of the high-tidal and supratidal lithologies (whether they are vegetated and replete with plant root bioturbation or are mud-cracked and have generated mud chips or contain evaporite minerals) and if organic matter-enriched sediment and/or peat forms the upper part of the stratigraphy. Climate also determines the nature of biota that colonize deltaic environments. The composition of these biota varies according to biogeographic setting, the occurrence of the biota relative to supratidal, tidal zones, and the subtidal and substrate type. The biota influence sedimentation and generation of lithotypes through shell and test production, root-structuring by trees, sedges, rushes and other salt marsh plants, bioturbation by plants and animals, and production of organic matter. Shell material contributes to the gravel and sand fraction in sediments and, through winnowing during wave action and storms, may be concentrated into sheets, lenses, and cheniers in mud-dominated sediments. The range of biota that directly contributes material to the lithotype or alter sedimentary structures by bioturbation and root-structuring include mangroves in tidal tropical environments, rushes, sedges, samphire, and other salt marsh plants in tidal tropical and subtropical environments, various crustacean-polychaete-mollusc assemblages in tidal tropical and subtropical environments, and wetland forests, sedges, and grasses in subaerial deltaic environments. Biota in intra-estuarine deltas are also described in Semeniuk and Semeniuk (this volume) in their description of Estuarine Deltaic Wetlands.

The stratigraphy of deltas within estuaries

Depending on whether deltaic sediments are dominated by sand, mud, mixtures of these sediment types, or gravel and whether the delta is fluvial dominated, wave dominated, or tide dominated, intra-estuarine deltas can exhibit a wide variety of stratigraphic types. Descriptions of the stratigraphy of intra-estuarine deltas are provided in Corner et al. (1990), Semeniuk and Semeniuk (1990a), Semeniuk and Semeniuk (1990b), Dalrymple et al. (1992), Allen and Posamentier (1993), Semeniuk (2000), and Semeniuk et al. (2011). A range of stratigraphic types in various hydrodynamic settings and with various contributions of sediment types is listed below and illustrated in Figures 6 and 7.



Deltas, Figure 7 Some case examples of the gross stratigraphy of fluvial-dominated, wave-dominated, or tide-dominated intra-estuarine deltas from southwestern Australia showing array of generalized lithology in terms of sand, muddy sand, and mud (Semeniuk and Semeniuk, 1990a; Semeniuk and Semeniuk, 1990b; Semeniuk, 2000; and Semeniuk et al., 2011).

Figure 6 shows a simplified and idealized gross-shoaled stratigraphy of deltas formed in sand-dominated, mixed sand-and-mud, and mud-dominated settings under hydrodynamics conditions of fluvial dominated, wave dominated, or tide dominated. The complications in stratigraphy due to lateral deltaic morphologic variation are not shown here. The sand-dominated deltas need not be exclusively sandy but may have a minor component of mud, and similarly, mud-dominated deltas need not be exclusively muddy but may have a minor component of sand. Sand in mud-dominated deltas can be exogenic (riverine sources) or endogenic (generated biogenically).

Microstratigraphic details of the various facies and subfacies of deltas in these different hydrodynamic environments can be found in Dalrymple et al. (1992), Allen and Posamentier (1993), and Semeniuk (this volume on “Stratigraphy of Estuaries”).

The sand-dominated delta developed under fluvial-dominated conditions is a wedge of sand prograded into the estuary. The sand-dominated delta developed under wave-dominated conditions is a wedge of sand comprising sediments of the beach-to-beach ridges, stacked and prograded into the estuary. The sand-dominated delta developed under tide-dominated conditions is a sequence of tide-aligned low-tidal to mid-tidal sand shoals, bars, and lenses that have shoaled to the level of high tide and that have been covered by floodplain sand deposits.

The sand-and-mud-dominated delta developed under fluvial-dominated conditions is a wedge of sand overlain by muddy sand and in turn overlain by mud prograded into the estuary. The sand-and-mud-dominated delta developed under wave-dominated conditions is a sheet of mud and muddy sand deposited at levels below the prevailing wave base and capping by a wedge of sand of beach-to-beach ridges, stacked and prograded into the estuary. The sand-and-mud-dominated delta developed under tide-dominated conditions is a sequence of low-tidal to mid-tidal, tide-aligned sand shoals, bars, and lenses that have shoaled to the level of high tide progressing through lithologies of muddy sand and mud and finally capped by floodplain mud deposits.

The mud-dominated delta developed under fluvial-dominated conditions is a wedge of mud prograded into the estuary. The mud-dominated delta developed under (moderate) wave-dominated conditions (*i.e.*, prevailing waves hydrodynamically dominate over tides and river flow) is generally a wedge of mud prograded into the estuary, but with the wave action and intermittent storms, there is local concentration of exogenic sand and biogenic sand and gravel through winnowing. These coarser sediments find expression in sheets of muddy sand, lenses of sand, and in cheniers. Mud accumulates below the prevailing wave base. The mud-dominated delta developed under tide-dominated conditions is a sequence of low-tidal to mid-tidal tide-aligned mud shoals that have shoaled to the level of high tide to form mud sheets that have been covered subsequently by floodplain mud deposits.

Case studies of the gross stratigraphy of fluvial-dominated, wave-dominated, or tide-dominated intra-estuarine deltas from southwestern Australia demonstrating generalized lithology in terms of sand, muddy sand, and mud are illustrated in Figure 7. The fluvial-dominated Harvey River delta shows a finger of sand (the prograded fans of sand at the delta front) overlying prodelta muddy sand and levee deposits of mud. The fluvial-dominated Collie River delta shows a sheet of riverine sand overlying prodelta mud and a capping of finer sand that has developed by construction of cheniers. The wave-dominated Deep River delta shows a stratigraphy of delta front and prodelta subtidal muddy sand overlain by the sand of beaches and beach ridges, with muddy sand filling inter-beach ridge swales. The wave-dominated Murray River delta shows a stratigraphy of delta front and prodelta subtidal sand overlain by sand of beaches and beach ridges and bars, with muddy sand filling inter-beach ridge swales and lagoons in a prograded bar-and-lagoon sequence. The tide-dominated Preston River delta shows a stratigraphy of subtidal sand that has shoaled through muddy sand with a capping of low-tidal to high-tidal mud.

Discussion and conclusions

Intra-estuarine deltas (also termed "bayhead" deltas) are the fluvial deposits that accumulate where one or more rivers enter an estuary. As with deltas formed in open marine coastal environments, these deltas can be classified as to morphology based on the response of the riverine sedimentary deposits to hydrodynamic setting. As such, deltas developed by fluvial-dominated, wave-dominated, or tide-dominated conditions can be identified. However, unlike the open marine coastal environment where the hydrodynamic conditions are regionally more uniform, deltas in estuaries experience a diversity of hydrodynamic conditions in the one deltaic setting and across the estuary. This is particularly the case where the estuary is large and complex in shape and where there is a strong component of wind that directs surface currents and wind waves. The main factors determining the morphology and landforms of deltas within estuaries are (1) the seasonality and strength of river flow which determines hydrodynamic conditions and the supply of sediment; (2) the salinity of the estuarine-receiving basin which determines the style of interchange of the river water with estuarine water (hypopycnal flow *versus* homopycnal flow *versus* hyperpycnal flow), the style of sediment delivery into the estuary, and, to some extent, the shape of the delta; (3) the magnitude of tides and wind waves which, in concert with the magnitude of river flow, will determine whether the hydrodynamic conditions will be dominated by fluvial, wave, or tidal processes; (4) the shape of the estuary; and (5) where the river(s) is/are located. Relatively simple estuaries, *e.g.*, narrow linear valley tracts with a single river mouth, are subject to interactions of river flow, tidal flux, and wind waves, with the delta morphology reflecting the locally dominant hydrodynamic condition. Complex estuaries and estuaries with large fetch create conditions where complicated hydrodynamics of waves and wind-induced currents interact with the shores of the estuaries and act on the deposits of the river or rivers entering the estuary, each river potentially being subject to differing hydrodynamics.

The morphology and landforms of intra-estuarine deltas respond to fluvial, wave, and tidal conditions. Fluvial-dominated deltas can be lobate, fan-shaped, elongate, or digitate. Wave-dominated deltas can be a lobate complex of prograded beach ridges, a series of beach ridges and/or recurved spits with intervening swales and/or linear lagoons, or a prograded series of bars and linear, oval to circular lagoons. The delta front of tide-dominated deltas usually is a crenulate to palmate complex of prograded subaqueous to tidal shoals, and the landward part of the delta is a floodplain. Complexity of local hydrodynamics may result in an intra-estuarine delta with different morphologies in different parts of the delta or contrasting landforms reflective of hydrodynamic conditions in a specific part of that delta.

Stratigraphy of intra-estuarine deltas is variable from delta to delta within the one estuary and variable between deltas in different estuaries because of the sediment types available and the hydrodynamic setting of the delta. The richness and variety of stratigraphic types in intra-estuarine deltas are due to (1) the provenance of the contributing rivers and types of sediment entering the estuary, (2) seasonality and strength of river flow, (3) tides, (4) wind and wind waves, and (5) climate. Climate plays a part in the development of delta stratigraphy in that rainfall and evaporation determine the nature of the high-tidal and supratidal lithologies and can determine if organic matter-enriched sediment and/or peat forms the upper part of the stratigraphy. Climate also determines the nature of biota that contributes shelly material as gravel and sand.

Depending on whether deltaic sediments are dominated by sand, or mud, or mixtures of sand and mud, or contain gravel, and whether the delta is fluvial dominated, wave dominated, or tide dominated, intra-estuarine deltas exhibit a variety of stratigraphic types. Sand-dominated deltas in fluvial-dominated conditions comprise a wedge of sand, while under wave-dominated conditions comprise a wedge of sand of beach-to-beach ridges stacked and prograded into the estuary, and those formed under tide-dominated conditions show a sequence of tide-aligned sand shoals, bars, and lenses aggraded to the level of high tide and that have been covered by floodplain sand deposits. The sand-and-mud-dominated deltas in fluvial-dominated settings comprise a wedge of sand overlain by muddy sand and in turn overlain by mud, while in wave-dominated settings are a sheet of mud and muddy sand deposited below the prevailing wave base with a capping of sand of beach-to-beach ridges stacked and prograded into the estuary. Those developed in tide-dominated settings show a sequence of tide-aligned sand shoals, bars, and lenses shoaled to the level of high tide through lithologies of muddy sand and mud and covered by floodplain mud deposits. Mud-dominated deltas in fluvial-dominated conditions are a wedge of mud prograded into the estuary, while those developed in wave-dominated settings comprise a wedge of mud but with local sand and gravel in sheets of muddy sand, lenses of sand, and cheniers with mud accumulating below the prevailing wave base. Mud-dominated deltas in tide-dominated settings comprise a sequence of tide-aligned mud shoals that aggrade to the level of high tide as mud sheets and are covered by floodplain mud.

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Cross-references

Delta Plain
Estuarine Deltaic Wetlands
Estuarine Geomorphology
Sediment Erosion
Shoreline Changes
Species Zonation
Stratigraphy of Estuaries
Tidal Hydrodynamics

DENSITY STRATIFICATION

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Definition

Density stratification can be defined as the vertical distribution of water masses into separate, distinct horizontal layers as a result of differences in density. These differences can also be attributed to differences throughout the water layers in dissolved solids, temperature, or suspended solids.

Description

The *density stratification* is extremely sharp, so that pure freshwater and pure saltwater are vertically adjacent. Such conditions can increase as density increases with depth and then the greater the vertical gradient will be, resulting in higher stability of the stratification.

Variations in the distribution of ocean density control the large-scale movements of water masses, and are important features in the dynamics of ocean surface currents, and drive the circulation of estuaries (Kjerfve, 1979). The less dense freshwater has a tendency to remain primarily in the surface layers. In estuaries where the tidal range is small, the tidal energy is limited during neap tides, and the water column becomes stratified vertically because of denser bottom water and a less dense surface layer.

In the North Atlantic Gyre, there are four distinct water masses that resulted from *density stratification*, creating interconnected currents with different flow characteristics and temperature: North Equatorial Current (NEC), North Atlantic Current (NAC), Gulf Stream (GS), and Canary Current (CC) (Talley et al., 2011).

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Cross-references

Estuarine Circulation
Residual Circulation
River-Dominated Estuary
Tidal Hydrodynamics

DETERMINING GEOHERITAGE VALUES

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Definitions

Geoheritage. The heritage value assigned to features of a geological nature encompasses globally, nationally, statewide to regionally, and locally significant features of earth science that are intrinsically important or culturally important, offering information or insights into the evolution of the earth or into the history of earth science, or that can be used for research, teaching, or reference (Brocx, 2008). It encompasses the variety of rocks types, stratigraphy, structural geology, geomorphology, and hydrology and covers a large variety of processes and products across a wide range of scales, from global tectonics, mountain building, landscape evolution to local surface processes and products such as weathering, erosion and sedimentation, cliff faces, fossils sites and mineral localities, and, at the microscale, diagenesis and deformation.

Geoconservation. This term refers to an action that works toward the preservation of sites of geoheritage significance for heritage, science, or education purposes. It can encompass all important geological features from the regional scale to the individual crystal, involving specific sites (special sites), or ensembles of geological sites. A "specific site" is where a significant geological feature occurs in isolation or may have historical or cultural significance; these have been formally identified in the British Isles as (geological) site(s) of special scientific interest (SSSI) or regionally important geological/geomorphologic sites (RIGS) (Ellis et al., 1996).

Geodiversity. Geodiversity is the natural variety of geological, geomorphological, pedological, and hydrological features of a given area and geological processes forming them (Brocx and Semeniuk, 2007). Use of the term, which etymologically means "the diversity of geological features," should be applied only in a region-specific or site-specific sense, i.e., not as a synonym for geology.

Geosite. This is a term used to denote small sites of geoheritage significance used for education, science, geotours, and reference.

Geopark. Geopark is used to denote large sites of geoheritage significance, usually an ensemble of geosites used for education, science, geotours, and reference.

Introduction

Estuaries stand as a distinct environment along the coast in that they bridge the aquatic hydrochemical environmental gap between freshwater and seawater. They can bring another aspect in addition to this hydrochemical setting because the landscape and geomorphic/sedimentologic setting of an estuary provide variability to the "mixing bowl" where freshwater and seawater interact. The emphasis on the landscape and geomorphic/sedimentologic settings of estuaries has resulted in their being classified according to a geomorphic framework or according to their origin (Fairbridge, 1980; Nichols and Biggs, 1985; Perillo, 1995). In this context, with their geologic, geomorphic, and sedimentologic characteristics and variability, they fall into the realm of geodiversity and geoheritage. As such, estuaries, in addition to the complexities and variability of styles of hydrochemical mixing, which is their first tier criterion of identification, provide a rich assortment of geologic, geomorphic, sedimentologic, mineralogic, and biogenic attributes such as shell deposits and bioturbation structures, not only in regard to the features within the estuaries themselves but also in the geology, geomorphology, and hydrology of the immediately surrounding landscape that frames or that has built them. Consequently, they hold potential to contain features and sites of geological significance or geoheritage value and lend themselves to qualifying as sites of geoheritage significance. This is especially the case in that estuaries, as sedimentary repositories, reside in various types of geologic and geomorphic settings, from rias to coastal plains to structural controlled (Fairbridge, 1980; Perillo, 1995), which results in a variety of geomorphic and sedimentologic estuarine types, and occur in a wide range of climates from tropical to temperate and from humid to arid, which also results in a variety of geologic/geomorphic, sedimentologic, and geochemical/mineralogic expressions. A large diversity (or geodiversity) of estuary types can be expected therefore from the perspective of the earth sciences.

Before a description is provided of the procedure to determine geoheritage values, the terms "geoheritage," "geoconservation," "geodiversity," "geosite," and "geoparks" are defined. They are associated with the concept of geoheritage, the enactment of geoconservation, and the inscription of geosites/geoparks.

Identifying and assessing sites of geoheritage value using the geoheritage tool kit

Identifying estuaries in different geological regions and the geoheritage essentials (i.e., the geological features that characterize an area) of these estuaries provides the first step in identifying sites for geoheritage. Clearly not all aspects of estuaries on earth are present in the one region, and not all aspects of an estuary in a region may be of geoheritage significance – the former recognizes the uniqueness, rarity, or representativeness of some estuarine features, and the latter requires some measure of assessment of significance. There are a number of ways that sites of geoheritage significance may be identified, and the British and European literature provides a history of how this has been achieved, with the final outcome being an inventory-based approach (Doyle et al., 1994; Wimbledon et al., 1995; Wimbledon, 1996; ProGEO, 2002; for discussion see Brocx, 2008).

The British and European approach to compiling an inventory of features of significance in the realm of geology has been successful in that numerous and varied aspects of geology have been identified and secured, but the approach has been thematic within a context of known geology and nationally specific geology. This is largely because the geology of European countries is reasonably well known, and these countries are relatively small (compared to, say, Australia, an island continent with a surface are of ~7.7 million km^2, and Africa with a surface area of ~30 million km^2). Australia has its own geological history with a vast array of geological features, from Archaean terranes to Proterozoic rock systems to Phanerozoic stratigraphy, lithology, paleontology, mineralization, etc., representing a wide diversity of processes and products developed under igneous, sedimentary, metamorphic, pedogenic, metallogenic, hydrologic, and diagenetic conditions. As such, Australia's estuaries also present exceptional geodiversity, reflecting the range of their geologic, oceanographic, and climatic settings. Therefore, to provide the framework for a category-based inventory of sites of geoheritage significance, Brocx and Semeniuk (2009, 2011) developed the geoheritage tool kit, to systematically identify and categorize sites of geoheritage significance. This method has been adapted to determine the geoheritage values of estuaries. The geoheritage tool kit uses six steps to identify geological features across various geological regions and at various scales, assign geological sites to various categories of geoheritage, and assess their levels of significance, and case studies are used here to illustrate the diversity of Australia's estuaries (Figure 1; Brocx and Semeniuk, 2009).

Step 1 identifies geological regions, providing a natural boundary to the estuary being investigated in terms of geological and geoheritage features, and an indication of the types of geological features that may be expected. It also ensures that comparisons in assessing significance are undertaken wholly within similar regions. Figure 2, for instance, shows the main regions of estuaries in Australia.

Step 2 identifies the geological essentials of a region and requires listing those geological features that *characterize* or are *peculiar* to a given natural region. For an estuary, it involves listing aspects such as the geological setting, estuary type, effects of climate, oceanography, and tidal range and interior features (such as flood-tidal deltas, shoals, tidal flats, deltas, basin type) and small-scale features (such as mineral precipitations, bioturbation types, and unusual or distinct sedimentary structures). The geological essentials of a region can be identified by drawing on the literature, interviewing scientists and, after identifying gaps in information, systematically obtaining further information from fieldwork. The list is termed the "geoheritage essentials" of an area.

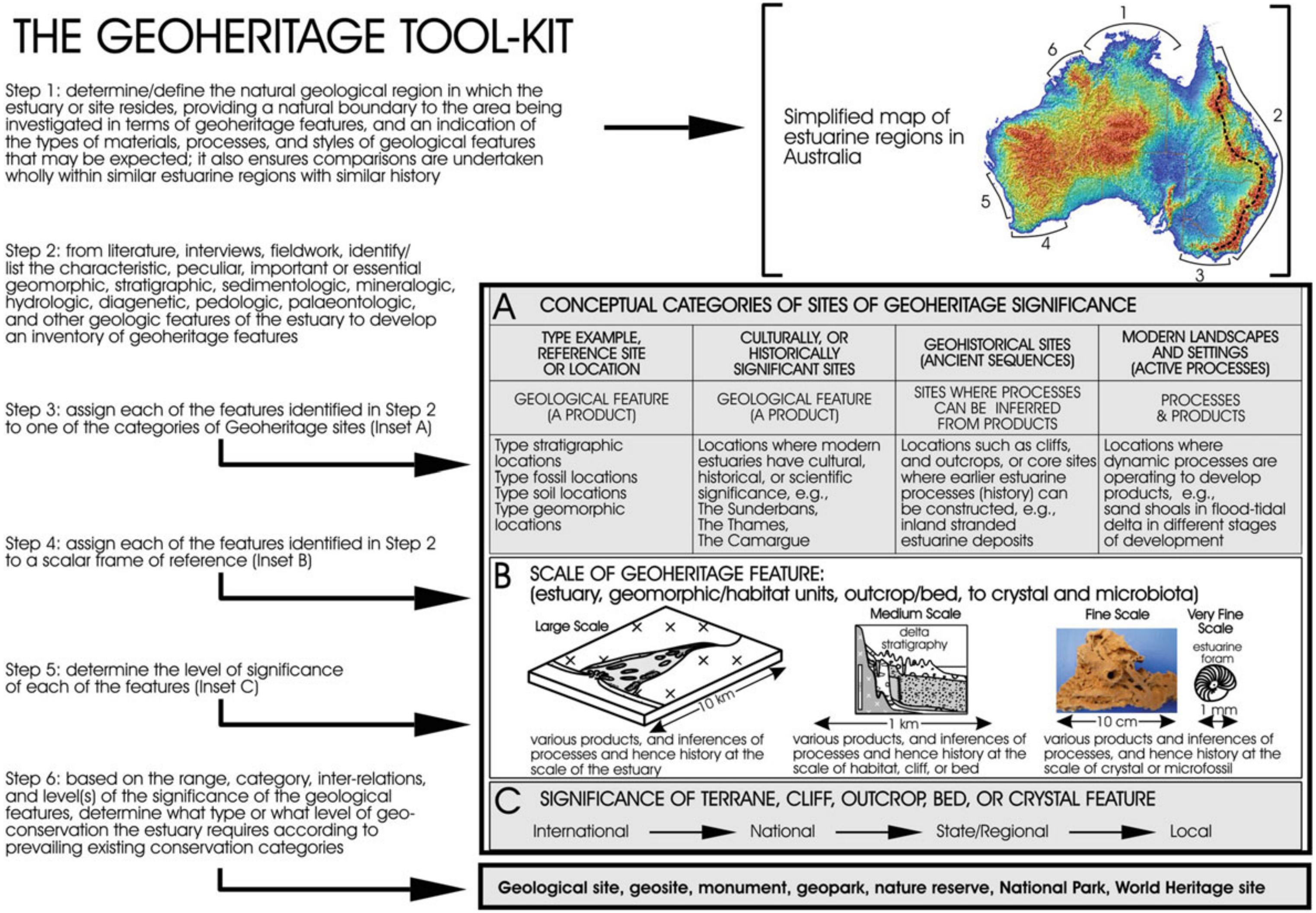
Step 3 allocates each unit of the inventory to a category of geoheritage, viz., a reference site, cultural site, geohistorical site, or a modern active landscape, so that comparisons in assessing significance are undertaken within similar categories. In regard to reference sites and/or type locations, once estuaries have been classified as to a type, the reference locations of end-member type or best example of an estuary can be identified and allocated as an international or national heritage locality. In this context, for comparisons of estuaries for geoheritage evaluation, it is important to have a worldwide applicable estuarine classification and nomenclature scheme that can be used systematically and comparatively to differentiate types based on landform/coastal setting, climate, shape and size of estuary, tidal and wave regime, sediment assemblages, seawater/ freshwater mixing style, and biota. A selection of estuaries that stand out globally as distinct and geomorphically significant because of either their size, internal landforms, representativeness, or naturalness and that could be used as estuarine reference sites and/or type locations are Lake St. Lucia (Natal, South Africa), Solway Firth (Scotland), Gironde Estuary (France), the Elbe (Germany), the deltaic complex of the Ganges-Brahmaputra (Sundarbans National Park, India), Walpole-Nornalup Inlet Estuary (Western Australia), Fitzroy River Estuary (Queensland), Gulf of Saint Lawrence (Canada), Chesapeake Bay (North America), and the Amazon Estuary (Brazil).

In regard to cultural or historical significance, estuaries may function as highly significant systems or may carry historical significance. The Camargue in the estuary of the Rhone (France) is an example of the former, and Port Hacking (Australia) and the Thames (the United Kingdom) are examples of the latter. Estuaries also can function as geohistorical sites showing ancient sequences where earth history can be determined. In regard to their stratigraphy and stranded geomorphology, they retain records over the past 7,000 years when sea level stabilized to its present position of coastal history and valley-fill history (Roy et al., 1980). Estuarine sequences that record estuarine evolution in Australia and North America (Fisher, 1969; Roy et al., 1980; Semeniuk, 2000) provide examples of the geohistorical importance of estuaries and illustrate the variety of pathways an estuary may take in its development. Estuaries also retain records of previous estuarine history in their stratigraphy and older estuarine geomorphology.

Estuaries illustrate modern landscapes and settings where earth processes are still active. They represent an environment where fluvial sedimentation interacts with basin processes to mobilize and deposit sediments into shoals, platforms, and basin-fill sheets. Flood and ebb tides form flood and ebb tidal deltas, and a plethora of biological, geochemical, hydrochemical, and physical processes at the finest scale result in various sedimentary deposits, biogenic deposits, sedimentary bedforms and structures, and mineral precipitates.

Some estuaries may belong to more than one geoheritage category. For instance, as a World Heritage area, the estuary of the Ganges-Brahmaputra river system serves as a reference site and as a location of modern landscapes and settings where estuarine and deltaic earth processes are still active in the largest tidal-dominated system in the world.

Step 4 allocates the geologic features to a scale, so that comparative assessments of levels of significance can be undertaken within a similar scale. The various scales



THE GEOHERITAGE TOOL-KIT

Step 1: determine/define the natural geological region in which the estuary or site resides, providing a natural boundary to the area being investigated in terms of geoheritage features, and an indication of the types of materials, processes, and styles of geological features that may be expected; it also ensures comparisons are undertaken wholly within similar estuarine regions with similar history

Step 2: from literature, interviews, fieldwork, identify/list the characteristic, peculiar, important or essential geomorphic, stratigraphic, sedimentologic, mineralogic, hydrologic, diagenetic, pedologic, palaeontologic, and other geologic features of the estuary to develop an inventory of geoheritage features

Step 3: assign each of the features identified in Step 2 to one of the categories of Geoheritage sites (Inset A)

Step 4: assign each of the features identified in Step 2 to a scalar frame of reference (Inset B)

Step 5: determine the level of significance of each of the features (Inset C)

Step 6: based on the range, category, inter-relations, and level(s) of the significance of the geological features, determine what type or what level of geo-conservation the estuary requires according to prevailing existing conservation categories

Simplified map of estuarine regions in Australia

A CONCEPTUAL CATEGORIES OF SITES OF GEOHERITAGE SIGNIFICANCE

TYPE EXAMPLE, REFERENCE SITE OR LOCATION	CULTURALLY, OR HISTORICALLY SIGNIFICANT SITES	GEOHISTORICAL SITES (ANCIENT SEQUENCES)	MODERN LANDSCAPES AND SETTINGS (ACTIVE PROCESSES)
GEOLOGICAL FEATURE (A PRODUCT)	GEOLOGICAL FEATURE (A PRODUCT)	SITES WHERE PROCESSES CAN BE INFERRED FROM PRODUCTS	PROCESSES & PRODUCTS
Type stratigraphic locations Type fossil locations Type soil locations Type geomorphic locations	Locations where modern estuaries have cultural, historical, or scientific significance, e.g., The Sunderbans, The Thames, The Camargue	Locations such as cliffs, and outcrops, or core sites where earlier estuarine processes (history) can be constructed, e.g., inland stranded estuarine deposits	Locations where dynamic processes are operating to develop products, e.g., sand shoals in flood-tidal delta in different stages of development

B SCALE OF GEOHERITAGE FEATURE: (estuary, geomorphic/habitat units, outcrop/bed, to crystal and microbiota)

various products, and inferences of processes and hence history at the scale of the estuary

various products, and inferences of processes and hence history at the scale of habitat, cliff, or bed

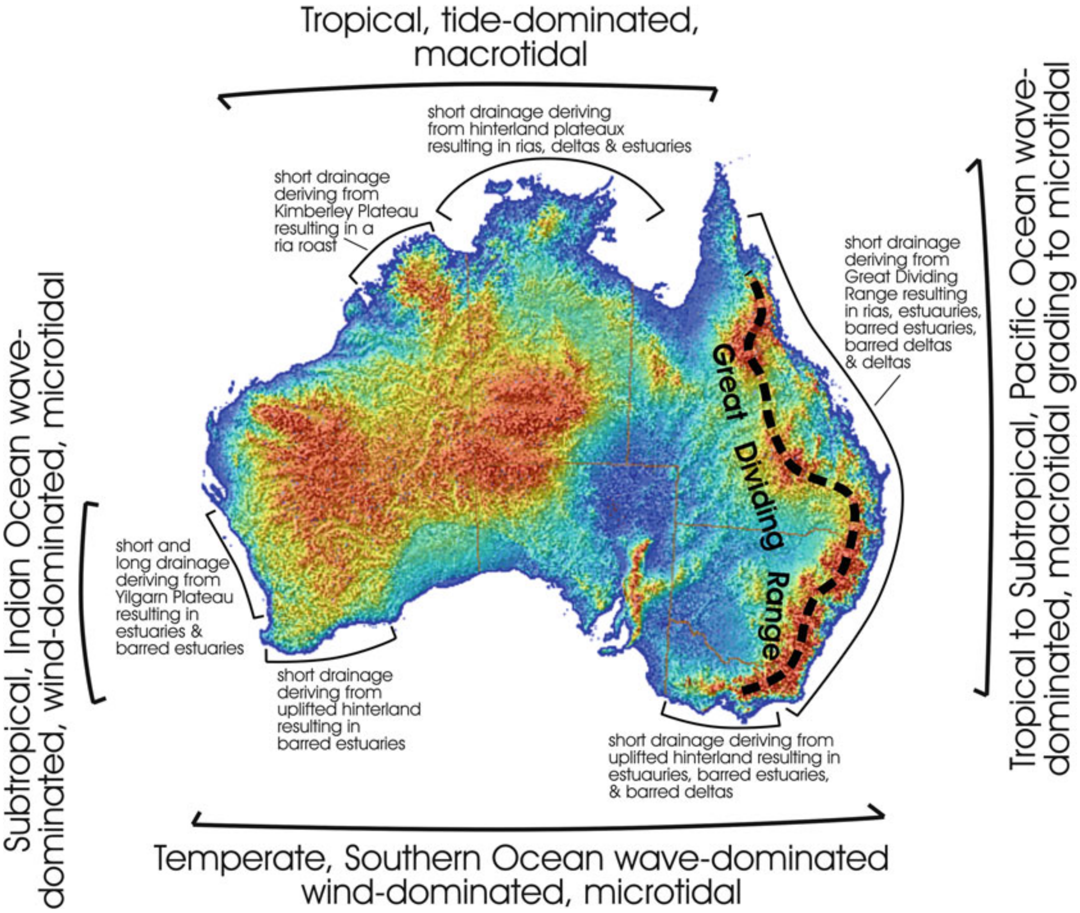
various products and inferences of processes, and hence history at the scale of crystal or microfossil

C SIGNIFICANCE OF TERRANE, CLIFF, OUTCROP, BED, OR CRYSTAL FEATURE

International → National → State/Regional → Local

Geological site, geosite, monument, geopark, nature reserve, National Park, World Heritage site

Determining Geoheritage Values, Figure 1 The six steps in the use of the geoheritage tool kit to identify geological features across various regions leading to designation of types of geoconservation. The *boxed* text and illustrations labeled *A*, *B*, *C*, from Brocx (2008), summarize the scope of geoheritage in terms of its categories, scales of application, and potential levels of significance that can be assigned to geosites.



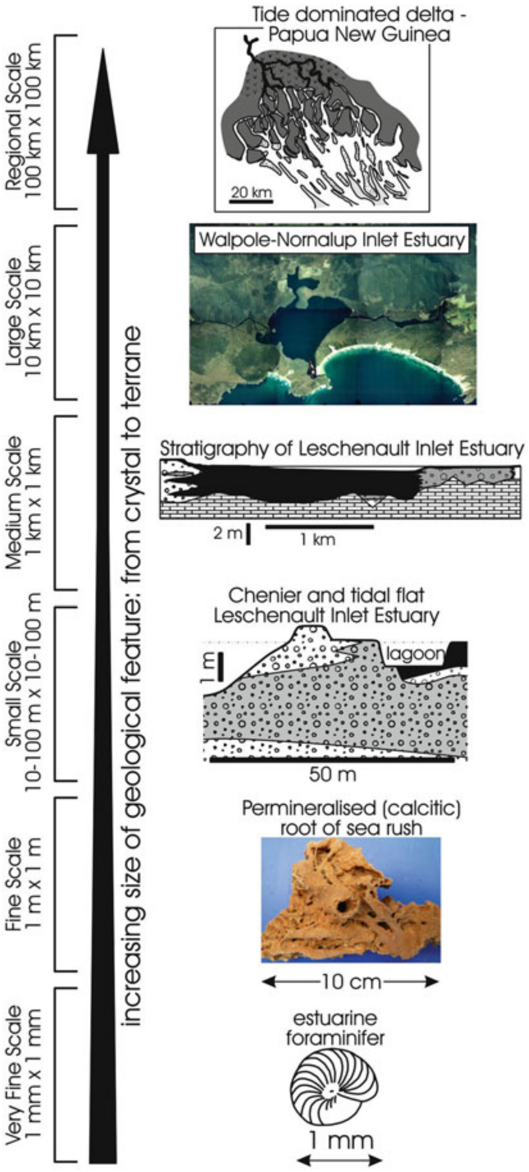
Determining Geoheritage Values, Figure 2 Simplified map of the main regions of estuaries in Australia based on geological setting, oceanography and tidal regime, and climate.

used in dealing with sites of geoheritage significance are regional, large, medium, small, fine, and very fine scales (Figure 3). Scale is important to consider in geoheritage/geoconservation since features of significance can range from crystals, bedding planes, and outcrops to that of landscapes and mountains. In many locations, sites are important because of crystal-sized phenomena and crystal fabrics (e.g., dolomite or permineralization in estuarine shorelines) or because of outcrops and bedding scale features (such as elevated estuarine fossil deposits). In the case of estuaries, the largest scale involves the size of the estuarine embayment, which may be tens of kilometers in size, and can involve the lower reaches of the drainage basin.

Step 5 assesses the level of geoheritage significance of the geological features regardless of their scale (Figure 4). The level of importance attributed to a given feature of geoheritage significance is related to how frequent or common is the feature within a scale of reference and/or how important is the feature to a given culture. Levels of significance are (Brocx and Semeniuk, 2007) (1) international, (2) national, (3) statewide to regional, and (4) local. Levels of significance of geoheritage features of (and within) estuaries are illustrated in Figure 4.

After an assessment of the range, categories, interrelationships, and level(s) of significance of the geological features, the final step is Step 6 which will determine what type and what level of geoconservation are assigned to the estuary whether in toto or in part.

Large estuaries, or sites within estuaries that are of geoheritage significance, or an amalgamation of numerous smaller sites of geoheritage significance can be assigned to geopark status. The Global Geoparks initiative supported by UNESCO sees geoparks as a territory encompassing one or more sites of scientific importance, not only for geological reasons but also by virtue of its archaeological, ecological, or cultural value. An estuary thus can qualify for this designation. The European Geoparks Network, established in 2000 (Zouros, 2000), defines a geopark as an area to conserve and valorize



Determining Geoheritage Values, Figure 3 Scales of features of geoheritage significance in estuaries.

geological heritage through the integrated and sustainable development of their territories. Similarly, an estuary can qualify also for this designation. The Asia Pacific Geoparks Network, founded in 2007, defines geoparks as nationally protected areas containing a number of geological heritage sites of particular importance, rarity or aesthetic appeal. These earth heritage sites are part of an integrated concept of protection, education, and sustainable development. An estuary can qualify also for this designation. All these initiatives aim to protect geodiversity, promote geological heritage, and support local sustainable economic development, thus involving community and commercial interests.

Estuaries lend themselves to designation as geoparks because they inherently have multiple uses (fishing, boating, shoreline nature walks, areas of conservation for waterbirds) and often illustrate interrelated features of landscape, geology, estuarine geomorphology, and sedimentology that can be utilized for science and education and tourism. Brocx and Semeniuk (2009) identified the Walpole-Nornalup Inlet Estuary in Western Australia as an integrated geopark, wherein the various Cenozoic and Holocene geological and estuarine features could be used as features for nature tours. Thus, estuaries can be viewed as potential geoparks, i.e., conservation, and promotional entities focused on geological and geomorphological attributes for local sustainable development.

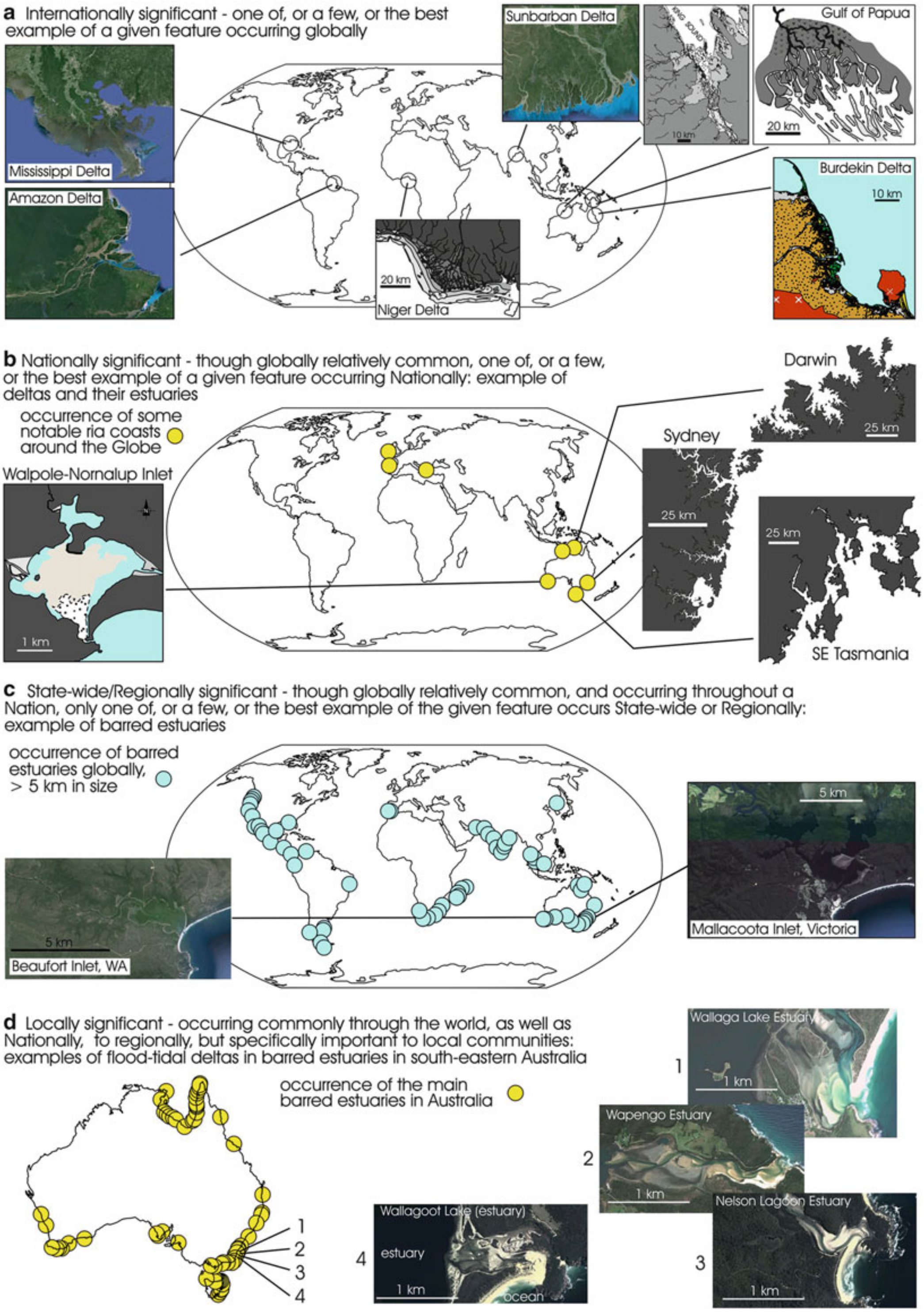
The leschenault inlet estuary and walpole-nornalup inlet estuary: case studies

The Leschenault Inlet Estuary and Walpole-Nornalup Inlet Estuary provide examples of the application of the geoheritage tool kit to identify and assess features of geoheritage significance in the estuaries. Both present two extremes of types in Western Australia. The Leschenault Inlet Estuary, a barrier dune barred estuarine lagoon, with two contributing rivers at its southern end, is located in a subhumid part of Western Australia, facing the swell-dominated Indian Ocean (Brocx and Semeniuk, 2011). Brocx and Semeniuk (2011) identify 10 features of geoheritage significance in the estuary. Of these, one feature is assessed as internationally significant, two as nationally significant, and seven as being of statewide or regionally significant (Figure 5). Brocx and Semeniuk (2011) proffer that the estuarine system, with its geological framework, complex shores, estuarine geomorphology and stratigraphy, and multitude of important small-scale features, also could function as geopark for geotours, research, and education.

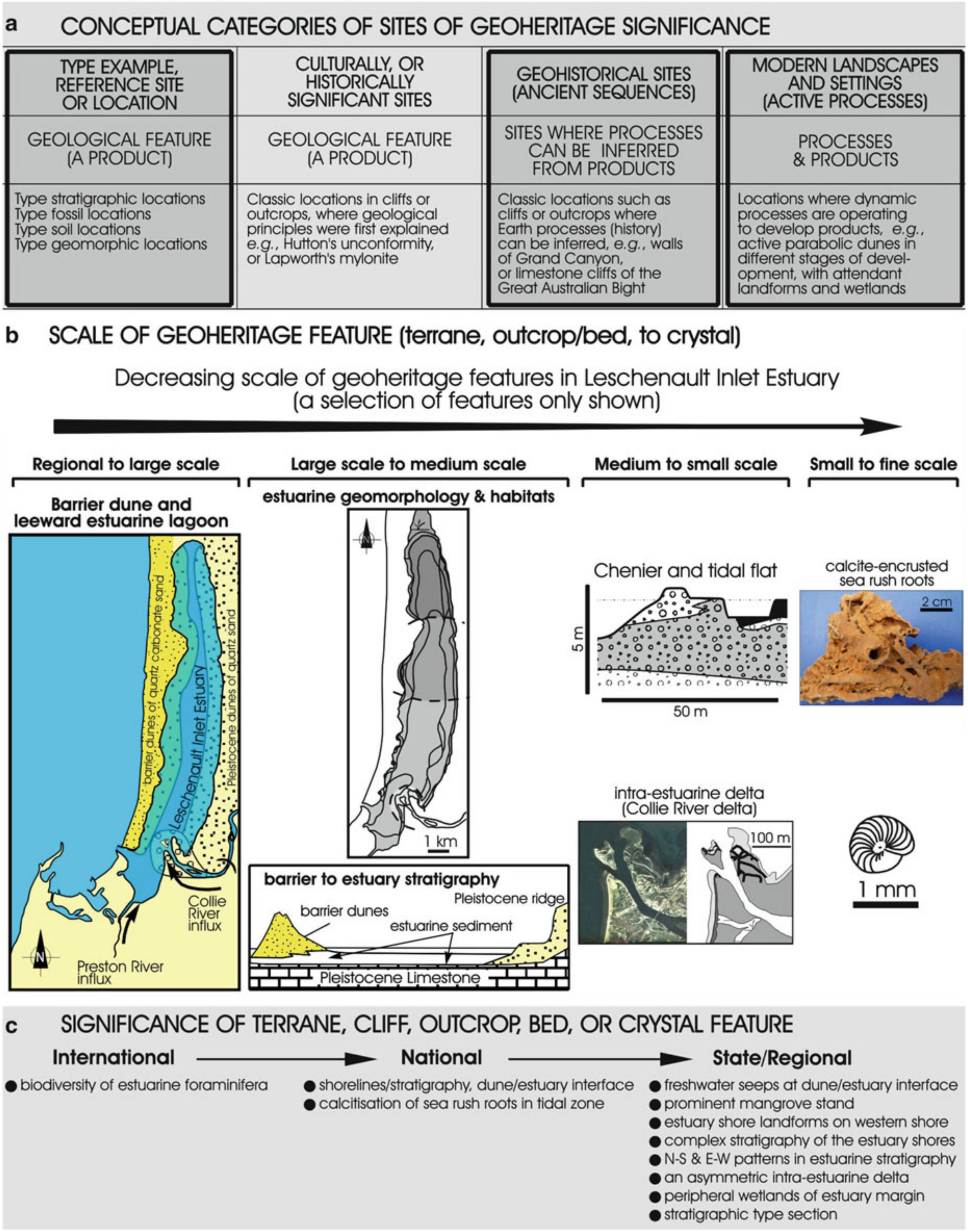
The Walpole-Nornalup Inlet Estuary, a twin ria estuary with a dune barrier and three contributing rivers, is located in the most humid part of Western Australia, facing the high-energy Southern Ocean (Semeniuk et al., 2011). Semeniuk et al. (2011) identify 22 features of geoheritage significance that include its intra-estuarine delta and peat-floored peripheral wetlands. Of these, one feature is assessed as internationally significant, two as nationally significant, and 19 as statewide or regionally significant (Figure 6). Semeniuk et al. (2011) proffer that the estuarine system of Walpole-Nornalup Inlet, with its geological framework, estuarine geomorphology and stratigraphy, and multitude of important small-scale features, could function as geopark for geotours, research, and education.

Summary

In estuaries, to date, there has been emphasis on their biological significance, e.g., their vegetation complexes, productivity, invertebrate fauna, and fisheries, and hence their conservation and management from a biological perspective, and less on the importance of their geology, geomorphology, sedimentology, hydrology, and geohistorical



Determining Geoheritage Values, Figure 4 Assessment of the level of geoheritage significance (based on the approach described by Brocx and Semeniuk 2007 but modified to focus on estuaries and deltas). The examples used to illustrate nationally significant geoheritage features are ria coasts of Tasmania, New South Wales, and Northern Territory (from Australia) and shown in their global context. The examples used to illustrate statewide/regional significance are barred estuaries (>5 km in size), drawn from Australia and shown in a comparative global and national context. The examples used to illustrate local significance are flood-tidal deltas from barred estuaries of southeastern Australia shown in a context of barred estuaries in Australia (See also "*Geoheritage*").



a CONCEPTUAL CATEGORIES OF SITES OF GEOHERITAGE SIGNIFICANCE

TYPE EXAMPLE, REFERENCE SITE OR LOCATION	CULTURALLY, OR HISTORICALLY SIGNIFICANT SITES	GEOHISTORICAL SITES (ANCIENT SEQUENCES)	MODERN LANDSCAPES AND SETTINGS (ACTIVE PROCESSES)
GEOLOGICAL FEATURE (A PRODUCT)	GEOLOGICAL FEATURE (A PRODUCT)	SITES WHERE PROCESSES CAN BE INFERRED FROM PRODUCTS	PROCESSES & PRODUCTS
Type stratigraphic locations Type fossil locations Type soil locations Type geomorphic locations	Classic locations in cliffs or outcrops, where geological principles were first explained *e.g.*, Hutton's unconformity, or Lapworth's mylonite	Classic locations such as cliffs or outcrops where Earth processes (history) can be inferred, *e.g.*, walls of Grand Canyon, or limestone cliffs of the Great Australian Bight	Locations where dynamic processes are operating to develop products, *e.g.*, active parabolic dunes in different stages of development, with attendant landforms and wetlands

c SIGNIFICANCE OF TERRANE, CLIFF, OUTCROP, BED, OR CRYSTAL FEATURE

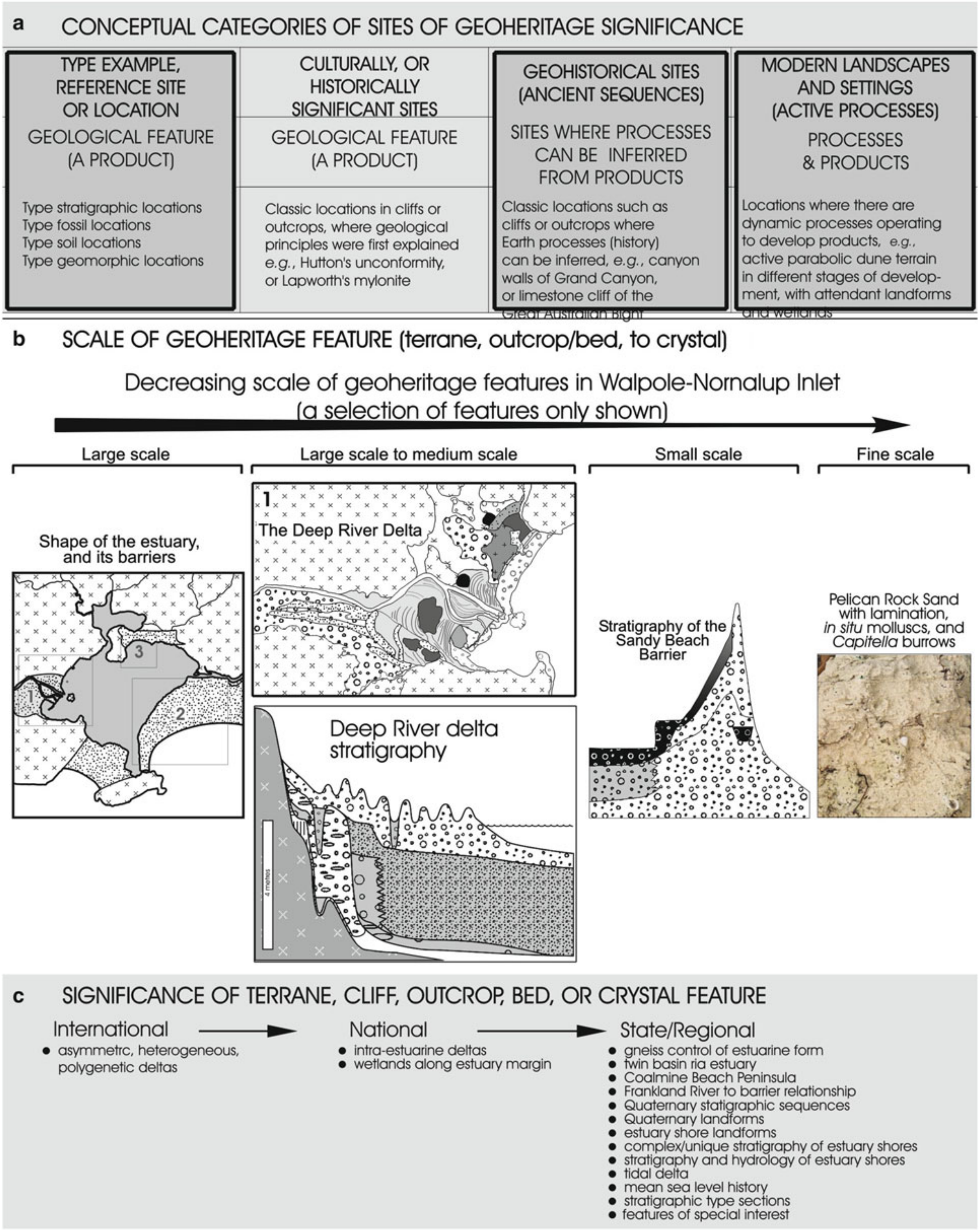
International → National → State/Regional

International:
- biodiversity of estuarine foraminifera

National:
- shorelines/stratigraphy, dune/estuary interface
- calcitisation of sea rush roots in tidal zone

State/Regional:
- freshwater seeps at dune/estuary interface
- prominent mangrove stand
- estuary shore landforms on western shore
- complex stratigraphy of the estuary shores
- N-S & E-W patterns in estuarine stratigraphy
- an asymmetric intra-estuarine delta
- peripheral wetlands of estuary margin
- stratigraphic type section

Determining Geoheritage Values, Figure 5 Application of the geoheritage tool kit to the Leschenault Inlet Estuary (Modified from Brocx and Semeniuk 2011). *Inset A* – the categories of geoheritage applicable to this area are highlighted in gray. *Inset B* – selected features of geoheritage significance are illustrated, graded in decreasing scale from *left* to *right* (a map of the barrier and lagoon, a map of estuarine habitats, cross section of barrier-to-lagoon stratigraphy, a chenier perched on a tidal flat, map of the Collie Delta, calcitized sea rush roots, and an estuarine foraminifer). *Inset C* – geoheritage features are allocated to a level of significance.



Determining Geoheritage Values, Figure 6 Application of the geoheritage tool kit to the Walpole-Nornalup Inlet area (Modified from Semeniuk et al. 2011). Most of the estuarine geoheritage features rank as regional to statewide significance, while some are national significance, and one feature of international significance.

evolution. However, there are two components to estuaries, i.e., the biotic and the abiotic (that underpins biodiversity). Geoheritage and geoconservation are concerned with the recognition and preservation of the abiotic world and in this context can be directed to the recognition and preservation of the geodiversity of estuaries. For instance, based on a world map of estuary types and their uniqueness or representativeness, it can involve the recognition and geoconservation of end-member types of estuaries as global "type examples" of the variety forms expressed around the world in response to climate, hydrodynamic setting, sediment types, and framework geology. At this scale, geoheritage recognizes the range of estuarine systems that are manifest around the world and attempts to address the significance of the variety of these estuaries that have formed in different geological, hydrological, sedimentological, and climatic settings within a variable biogeographic context. At the next level, geoheritage and geoconservation can involve the geoconservation of geological processes and products operating and occurring within estuaries, e.g., deltaic sedimentation and its variety of landforms, sand platforms and their surface bedforms, evolution of estuarine stratigraphy, stratigraphic/hydrologic interactions, and styles of hydrochemical mixing. At the finest scale, geoheritage and geoconservation can involve the recognition and geoconservation of microscale processes and products, often specific to an environmental setting and climate, e.g., diagenetic features such as calcitization of shoreline rush rhizomes, occurrence of dolomite, formation of pyrite nodules, the permineralization of skeletons, and the effects of freshwater seepage.

It should be noted that just as biologic systems are diverse, geological systems are also diverse (geodiversity), and in the case of estuaries, estuarine systems are also diverse and there are a large range of estuarine types, as exemplified by variation in their setting, shape, size, estuarine landforms, hydrology, and internal functioning. The classification of estuary types, using the geoheritage tool kit, has attempted to address this. Similar to the objective of nature conservation, to conserve the vast diversity of life forms, an objective of the conservation of sites of geoheritage significance in estuaries would be the conservation of the variety of their forms on the earth. In this context, the conservation of a single "estuary" as an example of an estuarine system as representative of the full variety of estuarine types globally is insufficient. If estuaries, for instance, exhibit a large diversity of geometric and hydrologic types, stratigraphic fills, and origins, then at the least their conservation should encompass an example of each of the types.

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Cross-references

Geoheritage

DETRITUS FOOD WEBS

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Synonyms

Decomposer food web food; Detritus cycle; Microbial loop

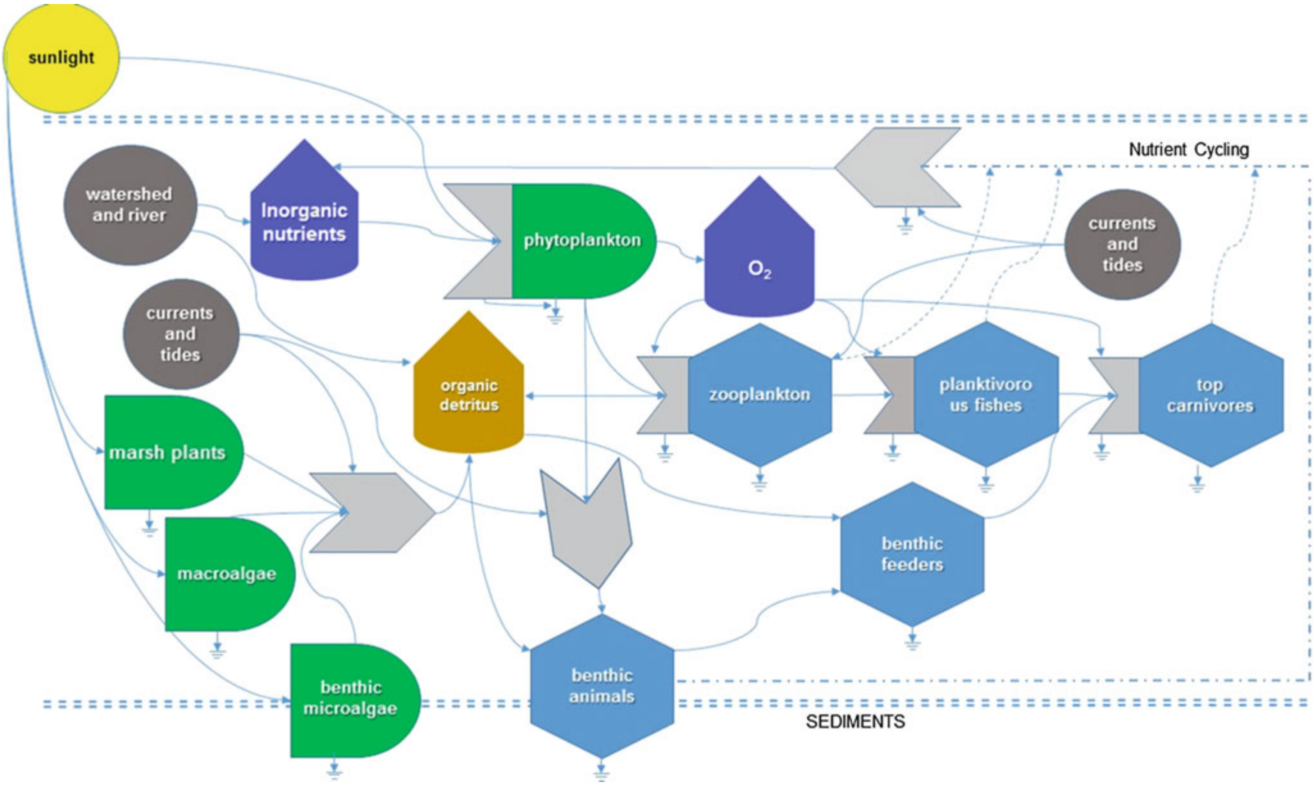
Definition

Food webs or portions thereof that are based on the decomposed particles of dead plants and animals, mediated by saprotrophic and scavenger organisms that break down organic matter into its constituent compounds.

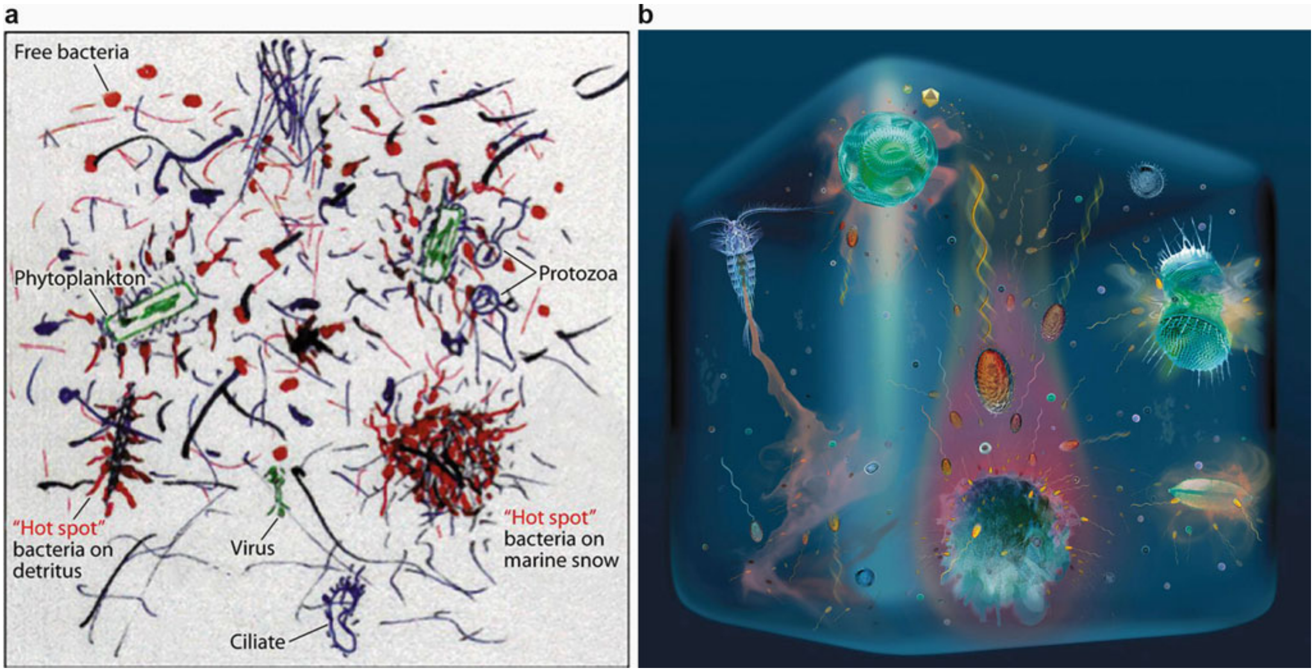
Summary

While the basic photosynthetic production processes supporting all but extremophile-based food webs do not differ among most ecosystems, from an energy flow perspective (see *Food Web/Trophic Dynamics*), the pathways whereby organic compounds reach metazoan consumer organisms can be both intricate and often confusing. This is especially the case in estuaries, wherein diverse living and detrital organic matter sources support mixed autotrophic and heterotrophic production (Figure 1). As opposed to direct herbivory ("grazing") of living plants, food webs based on detritus involve the decay of photosynthetic products and even dead consumer organisms; however, whether or not detritus should be defined as including associated living decomposers and other microorganisms (Figure 2) has always been somewhat of a philosophical dispute (Darnell, 1967).

While detritus is the predominant food web source in some ecosystems, such as in soils, the occurrence and contribution of detritus to aquatic food webs have been more debatable. About the same time that Sir Alistair Hardy (1924) was describing a food web that supported Atlantic herring wholly by autotrophic production from algae, Summerhayes and Elton (1923) diagrammed a "nitrogen cycle" for Bear Island (Bjørnøya), Svalbard, that illustrated a more complex network also involving detritus from aquatic and terrestrial plants being decomposed by bacteria and protozoa before sustaining detritivores and ultimately higher-level consumers. With the discovery of



Detritus Food Webs, Figure 1 Detritus food web. Estuarine food web energy flow indicating interactions among autotrophic (*green*: phytoplankton, marsh plants) and heterotrophic (*brown*: detritus) pathways of organic matter production, consumption (*blue*: consumers), inorganic resources (*purple*), transformations and storage (*light gray*) and energy (*dark gray*) (Modified from Day, J. W., Jr., B. C. Crump, W. M. Kemp and A. Yáñez-Arancibia (eds.) 2012. Estuarine Ecology, 2nd Edition, Hoboken, New Jersey: Wiley-Blackwell).



Detritus Food Webs, Figure 2 Illustration of the different forms of detritus and microbial microenvironments common to estuaries, including (left image; from Stocker and Seymour 2012 Microbiol. Mol. Biol. Rev. 76:792-812) "hot spots" of microbial activity in association with detritus, marine snow particles, and phytoplankton cells, and (right image; modified from the cover of Science, 5 February 2010; original image credits: R. Stocker, J. R. Seymour, G. Gorick) organic matter source, including zooplankton excretions (left), phytoplankton exudation (the "phycosphere") (top; bottom right), phytoplankton lysis (top right), settling marine snow particles (center bottom), and copepod excretions (left).

"marine snow" (Alldredge and Silver, 1988) and the attendant "microbial loop" driven by dissolved organic matter (POM) (Pomeroy, 1974; Azam et al., 1983), even presumed autotrophically dominated ocean food webs were found to have highly integrated detritus pathways (Figure 2b). While detritus has long been considered to be a major driver of food web pathways in estuarine sediments (e.g., Newell and Field, 1983), it also became even more relevant to estuaries overall (Crump et al., 2012), especially with increased understanding of gravitational circulation processes that promote estuarine turbidity maxima as "biogeochemical reactors" (Baross et al., 1994; Savoye et al., 2012). What has become increasingly obvious from the more recent application of isotope and other biomarker sampling and experimentation in estuaries is that although detritus fuels and may even dominate many estuarine food webs, the extent to which it does varies considerably as a function of the type and region of estuary and the time frame (Odum, 1984; Peterson et al., 1985; Peterson and Howarth, 1987; Deegan and Garritt, 1997; Akin and Winemiller, 2006).

In many respects, estuaries have often been the nexus of the debate about the role of detritus food webs, touching on the core of many fundamental issues in ecological theory such as labile versus refractory organic matter sources (Mann, 1988); the importance of allochthonous, spatial subsidies (Polis et al., 1997); outwelling (Childers et al., 2000); compartmentalization (Raffaelli and Hall, 1992); community stability (Huxel and McCann, 1998); and top-down versus bottom-up control on food web structure (Power, 1992). While the prominence of detritus in estuarine food webs is less debatable, its role in shaping estuarine ecosystem dynamics and regulating the productivity of important consumers such as commercial fisheries is still somewhat controversial.

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Cross-references

Food Chain
Food Web/Trophic Dynamics

DIAGENESIS

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Definition

The chemical and biological environment within sediments is very different from the overlying water column from which the particles settled. During burial, particles undergo diagenesis: the transformation of sediment and organic matter by physical, biological, and chemical processes. Early diagenesis refers to the transformations that occur while sediments are submerged, temperatures do not exceed 140 °C, and burial is less than a few 100 m (Berner, 1980).

Description

Physical processes alter sediments after deposition. Sediments are compacted by the weight of overlying sediments, which decreases the ratio of interstitial water to sediment. If oxygen is present in overlying water, benthic macrofauna will mix sediments. Bioturbation is most intense near the sediment-water interface and decreases with depth. In specific settings, soft sediments deformation structures can form, including dewatering structures, slumped beds, and load structures.

The chemical and biological environments change with distance from the sediment-water interface. Exchange between interstitial water and overlying water is restricted, allowing for the composition of interstitial water to differ from overlying water. Moving deeper into sediments, interstitial water becomes more reducing as oxidants are consumed during respiration (Froelich et al., 1979). Respiration also increases the acidity of interstitial water, reducing the pH. The composition of interstitial water is further influenced by uptake and release of compounds in biotic and abiotic reactions. These changes to interstitial water chemistry allow for different transformations of organic matter and sediments to occur.

Most organic matter deposited in sediments is removed by respiration of benthic organisms. However, some organic matter is transformed from characterized compounds, such as lipids, carbohydrates, and amino acids, into uncharacterized humic substances. This likely occurs biologically through the selective utilization of more reactive components of organic matter, with some contribution of abiotic recombination of smaller molecules (Burdige, 2007). Humic substances tend to be refractory, persisting for long periods in the sediment. The diagenesis of organic matter depends on the redox conditions, with greater preservation of humic substances under more reducing conditions.

Inorganic sediments also undergo diagenesis from a variety of mechanisms. Sediments may be transformed as they pass through the gut of detritus feeders. Mineral dissolution of carbonates and silica may occur. In anoxic sediments, oxidized minerals, such as Fe_2O_3 and MnO_4, can be removed by microbial respiration. Further, the loss of these high surface area minerals can greatly reduce the adsorption capabilities of the sediment. Adsorption and desorption reactions can also occur due to changing pH and Eh conditions. Ion exchange can occur in clays, altering their composition. Authigenic minerals, including phosphates, carbonates, and sulfides, may precipitate out of solution. Some precipitates can cement sediment grains, reducing the ratio of interstitial water to sediment.

During later stages of diagenesis that occur with greater burial, compaction and heating of sediments can lead to the loss of water from hydrous minerals, cementation of sediments, and lithification of sediments. As a result, the physical structure and chemical composition of buried sediments and sedimentary rocks depends both on the initial composition of the material deposited and the diagenesis that occurs during burial.

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Cross-references

Anoxia, Hypoxia, and Dead Zones
Sediment Toxicity

DIFFUSION

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Synonyms

Scatter

Definition

Diffusion is defined as free or random movement of molecules from a higher concentrated to lower concentrated region. Concentration gradients are part of this random movement.

Description

Lewis (1997) defined two criteria for molecular motion to be considered diffusion. First, the number of molecules moving in two directions (from high to low or vice versa) must be equal. The occurrence of an unbalanced condition is called advection. The second criterion is the occurrence of a concentration gradient between two regions.

Free molecular motion, as molecular diffusion, is described by Fick's law and diffusion equation. The movement of particles under turbulent motion can be defined as turbulent diffusion or eddy diffusion. The difference in turbulent diffusion is explained by the eddy diffusion coefficient (Fischer et al., 1979) (Figure 1).

For a one-dimensional case, diffusive transport can be expressed by using Fick's law as follows:

$$J = -D.\frac{\partial C}{\partial x} \tag{1}$$

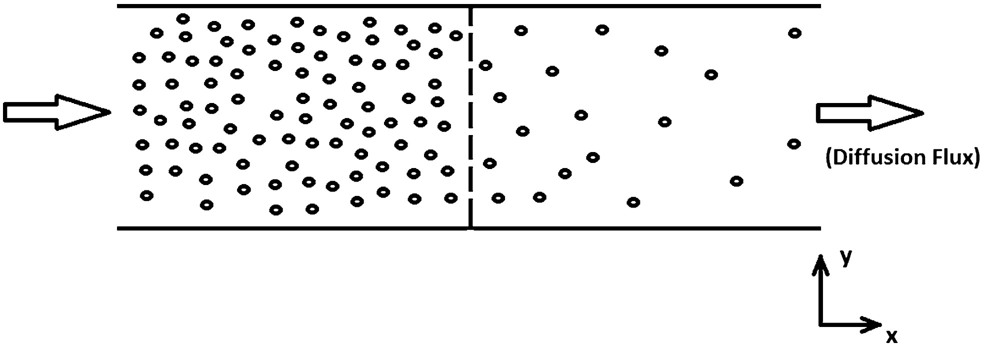
where J is the diffusion flux as the molecular amount of particles or substance per unit area per unit time (mol.m^{-2}.s^{-1}), D is the molecular diffusion coefficient (m^{2}.s^{-1}), and $\frac{\partial C}{\partial x}$ is the concentration gradient (mol.m^{-4}). The negative sign in the equation indicates the flux from high concentrated to less concentrated regions.

The turbulent diffusion (1) can be redefined as

$$J = -K.\frac{\partial C}{\partial x} \tag{2}$$

where K is the coefficient of eddy diffusion or turbulent diffusion.

The molecular diffusion coefficient can be taken as a constant at a defined temperature and also can be regarded as a property of the fluid. Conversely, the turbulent or eddy diffusion coefficient depends on the



Diffusion, Figure 1 Diffusion is the free or random movement of molecules from a higher concentrated to lower concentrated region.

Diffusion, Table 1 Molecular diffusion coefficients at infinite dilution in 25 °C water (Cussler, 2009)

Solute	Coefficient ($\times 10^{-5}$ cm^2/s)
Ammonia	1.64
Carbon dioxide	1.92
Hydrogen sulfide	1.41
Oxygen	2.10

strength and size range of the eddies in the turbulent motion, and it is not constant in all the fluid body (Lewis, 1997).

In general, the coefficient of eddy diffusion or turbulent diffusion (*K*) is a thousand times higher than the molecular diffusion coefficient (*D*) (Lewis, 1997). The diffusion coefficient affects the movement of particles or molecules (Table 1). The most common method for estimating diffusion coefficients for liquids uses the Stokes-Einstein equation.

Estuaries are semi-enclosed coastal bodies of water where freshwater mixes with saltwater and multiple factors affect the system hydrodynamics such as tides, currents, waves, Coriolis force, freshwater inflow, saltwater inflow, meteorological effects, and bathymetry (Dyer, 1973). Estuarine transport therefore is a complex process (Ambrose, 1990).

Mostly in estuaries, the primary mixing mechanism is not caused by the molecular viscosity or diffusion, but turbulent mixing. Turbulent eddies transfer a water body into other parcels having different mean velocities causing different water properties (Martin and McCutcheon, 1998).

Total mixing depends on diffusion which is the sum of molecular diffusion and turbulent or eddy diffusion, and coefficients can be summed ($D + K$). However, the molecular diffusion coefficients are considered negligible since they are so much smaller than the turbulent or eddy diffusion coefficients (Martin and McCutcheon, 1998).

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Cross-references

Dispersion
Tidal Hydrodynamics

DISPERSION

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Synonyms

Dissipation; Scattering

Definition

Dispersion in estuaries is the spreading or scattering of dissolved or suspended substances due to a combination of shear (or nonuniform velocity profile) and turbulent diffusion (Baretta-Bekker et al., 1995).

Description

The main difference between diffusion and dispersion is the longitudinally or laterally nonuniform velocity profile. Dispersion reflects the scattering of a cross-sectional mean concentration, whereas diffusion represents the scattering of a local concentration (Gulliver, 2012).

Dispersion coefficients have been determined for estuaries and other water bodies. These values have been compiled and listed in many publications. Dispersive mixing is not turbulent diffusion, but rather is due to nonuniformities in velocities and concentrations (Martin and McCutcheon, 1999). The collection of field data is very important for determining dispersion coefficients because many parameters in estuaries and other water bodies affect hydrodynamic mixing.

The fundamental papers on shear dispersion were published in the early 1950s by Geoffrey Ingram Taylor. His theoretical work applied to open channel flow (Elder, 1959) and to coastal waters (Bowles et al., 1958).

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Cross-references

Diffusion

DISSOLVED OXYGEN

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Synonyms

Elemental oxygen; Dioxygen; DO; O_2

Definition

Dissolved oxygen (DO) is the amount of elemental oxygen (chemical symbol O_2, molecular wt 31.99 g/mol) dissolved in fresh or salt waters.

Controlling factors of dissolved oxygen

The measurement of dissolved oxygen in water is provided in mg/L or ml/L units for environmental regulatory purposes but is usually measured in μMol for chemical and oceanographic studies. Table 1 provides conversions for these units. Most dissolved oxygen in estuarine waters is due to exchange with the atmosphere at the seawater surface. Atmospheric oxygen (O_2) presently constitutes 20.9 % of the atmosphere by volume and 23.1 % by mass. The maximum amount of DO at equilibrium with the atmosphere (100 % saturation) depends on the atmospheric pressure (partial pressure) of oxygen and the temperature and salinity of the water. As temperature and salinity increase, dissolved oxygen saturation decreases, while increases in atmospheric pressure increase saturation concentration. Additionally, photosynthesis by primary producers can increase surface water concentration to supersaturation levels, while aerobic respiration processes can decrease it to hypoxic levels at depth. Therefore, dissolved oxygen concentration is not conservative and is strongly affected by biotic organisms. Accurate calculation of the exact saturation value is a quite complex function of temperature, salinity, and pressure. Due to very slight discrepancies in results using the Weiss equations (Weiss, 1970; USGS, 1981), the United States Geological Survey (USGS) has changed saturation equations (USGS, 2011) using more recently published equations (Benson and Krause, 1984; Garcia and Gordon, 1992). The USGS revised its methodology in 2011 to follow the Benson and Krause equations. The USGS maintains a Web site that provides such calculations for saturation values at specific temperatures and salinities (USGS, 2013).

Measurement methodologies

The classic method to measure dissolved oxygen in water involves titration of treated water samples using the Winkler (iodometric) method (Winkler, 1888) and is considered one of the most accurate methods assuming all precautions are followed in the sampling procedures, handling and addition of reagents involved. The original method has been modified due to interference from nitrite, ferrous or ferric iron, and organic matter (Carpenter, 1965; Strickland and Parsons, 1968; APHA, 2005), while iodate may still cause problems (Wong and Li, 2009). Poor handling can expose water samples to gas bubbles during the initial addition of reagents to fix the sample in the field and introduce significant overestimate errors. The method is considered precise for lab analyses, but other methods are recommended for measurements in situ (Lewis, 2006). Because accurate Winkler measurements are difficult at extremely low DO levels, spectrophotometric methods using special dyes such as Rhodazine D are sometimes recommended for such situations (Broenkow and Cline, 1969; White et al., 1990; Lewis, 2006). The use of amperometric techniques for real-time field measurements has been accepted as a suitable method to determine in situ dissolved oxygen in fresh and salt waters as long as corrections based on temperature and salinity are made (usually provided within the instrumentation). This method requires careful calibration of the sampling device. The "Clark"-type amperometric method uses a silver (Ag) anode and a gold (Au), platinum (Pt), or palladium (Pd) cathode surrounded by an ionic fluid (usually KCl). A thin, gas permeable Teflon® membrane allows exchange of oxygen with the electrodes. Because the reaction at the electrode consumes oxygen, accurate membrane response requires flowing water to achieve steady equilibrium conditions, leading to a need for mixing or forced flow of the water being sampled across the membrane as well as time for equilibrium to be achieved. Another oxygen probe type (galvanic) has a self-polarizing amperometric cell that uses a lead (Pb) or zinc (Zn) anode and a gold (Au) or silver (Ag) cathode. An electrolyte of NaCl or NaOH surrounds the electrodes (Eutech Instruments Pte Ltd., 1997). If either of these sensors is deployed for long periods, overgrowth by biofilms and fouling organisms on the membrane can interfere with the gas exchange, so membrane replacement is required at certain intervals. Manufacturers recommend various antifouling techniques to decrease the rate of biofouling growth at the membrane. Anoxic waters with high levels of hydrogen

Dissolved Oxygen, Table 1 Conversions for various measures of dissolved oxygen at 100 % saturation at 760 mmHg; ρ is the density of the sample based on the equation of state (Unesco, 1981).

ml/L DO to mg/L	mg/L DO to ml/L	ml/L DO to μMol/L[a]	μMol/kg
ml/L DO * 1.42903 = mg/L	mg/L DO * 0.6998 = ml/L	ml/L DO * 44.660 = μMol/L	$\frac{uMol}{L}/\rho = \mu\text{Mol/kg}$

[a] Common oceanographic CTD instrumentation and others use this historic method to calculate uM/L, but exact measurements required more sophisticated calculations for exact uMol concentration (Thierry et al., 2011)

sulfide (H_2S) can “poison” the electrodes, decreasing the response to oxygen concentrations. In more recent years, a luminescent technique has become commercially available using a sensor called an optode with a membrane impregnated with a dye which emits red light frequencies when excited by a blue laser. Both the intensity and duration (lifetime) of the fluorescence signal are affected by temperature and are quenched by DO in a linear response at low to mid saturation levels. Because of this, temperature measurements of high precision are required. This method has a number of advantages, including less interference from H_2S and biofouling and greater sensitivity under low DO conditions since oxygen concentration decreases the fluorescence response, so the strongest signal occurs under anoxic conditions. However, at high saturation values, the response is more complex and requires a complex polynomial relationship between DO, temperature, and the fluorescence signal. The dye can degrade over time and so requires membrane replacement at set intervals (Mitchell, 2006; YSI, 2009). Some sensors measure the intensity, while others measure the lifetime of the emitted signal.

Summary

Dissolved oxygen (DO) is the amount of elemental oxygen (Chemical symbol O_2, molecular wt 31.99 g/mol) dissolved in fresh or salt waters. It is measured as mg/L or ml/L for environmental regulatory purposes but is usually measured in uMol for chemical and oceanographic studies. The maximum amount of DO in water at equilibrium with the atmosphere (100 % saturation) depends on the atmospheric pressure (partial pressure) of oxygen and the temperature and salinity of the water. As temperature and salinity increase, dissolved oxygen saturation decreases, while increases in atmospheric pressure increase saturation concentration. Dissolved oxygen concentration is not conservative and is strongly affected by biological processes such as photosynthesis and respiration.

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Cross-references

Aerobic Environments
Anaerobic Environments
Ecological Monitoring
Estuarine Total Ecosystem Metabolism
Eutrophication
Halocline
Microbial Degradation
Oxygen Depletion
Water Quality
Well-Mixed Estuary

DREDGE AND FILL

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Synonyms

Sediment disposal; Sediment excavation

Definition

Dredge-and-fill operations are conducted in coastal areas mainly to re-nourish beaches, to restore wetland habitat, to remove excessive amounts of bottom sediments from waterways, and to construct lagoons and roads. Dredging in estuaries is carried out to create new harbors, berths, and waterways or to improve navigation. “Large estuaries, such as Coos Bay, Oregon (U.S.), that function as deep-water ports for large freighters, tankers, and other ships require the deepest channels and most frequent channel maintenance” (Oberrecht, 2005).

Fill is an operation that is conducted mostly in coastal regions using sand, rocks, gravel, shell, earth, and concrete as filling materials. Filling activities in estuaries, lagoons, and coastal wetlands may include restoring and modifying areas by deposition of sediments.

Characteristics

Dredgers are used to excavate bottom sediments from estuarine water bodies that can then be dumped at appropriate locations. “Dredging is accomplished basically by two mechanisms: (1) hydraulic dredging – removal of loosely compacted materials by cutterheads, dustpans, hoppers, hydraulic pipeline, plain suction, and sidecasters, usually for maintenance dredging projects; and (2) mechanical dredging – removal of loose or hard compacted materials by clamshell, dipper, or ladder dredges, either for maintenance or new-work projects” (San Francisco Bay Conservation and Development Commission, 2001).

Dredge-and-fill operations may be deleterious or beneficial to certain species of organisms in estuaries and wetlands. Johnston (1981) noted that the ways to mitigate adverse effects of dredge-and-fill operations should include careful pre- and post-construction environmental studies. Dredge-and-fill activities are regulated in the USA by municipal, state, and federal government agencies.

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Cross-references

Dredging

DREDGING

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Definition

Dredging is the process of excavating bottom sediments from the estuarine floor for disposal at another location, most frequently to increase the depth of a channel to facilitate navigation by floating vessels.

Introduction

Early navigators were in many cases limited by naturally occurring depths in water bodies. As ships grew larger, dredging became necessary to increase water depths to allow safe passage. In some cases, dredging was used to create navigable water where land previously existed, with the Suez and Panama Canals serving as two prominent examples from the nineteenth and twentieth centuries, respectively. However, the digging of canals predates recorded history.

Tidal inlets connect rivers and estuaries to adjacent seas and are thus important for marine commerce. The natural depth within an inlet is typically controlled by a balance between tidal currents sweeping through, in alternating directions, and waves and longshore currents pushing sediment into the inlet. Dredging can effectively increase the depth in the channel. But without any changes in the tidal prism that defines the volume of flow through the inlet per tidal cycle, the channel is then deeper than its equilibrium configuration and subject to shoaling (van de Kreeke, 1992). This implies that dredging to greater depths will result in an increased need for maintenance dredging.

Dredging is also critical for maintenance of ship berthing areas and turning basins. It is widely employed for land reclamation purposes, habitat creation, sand mining, and beach nourishment activities. Project scope can range from maintenance of a small boat launching area up to major land reclamation projects involving many millions of cubic meters of sediment (e.g., Ports of Los Angeles in the United States and Rotterdam in the Netherlands; Palm Islands in Dubai). Bray and Cohen (2010) provide other examples of projects around the world.

Equipment and techniques

Dredging has been conducted by a wide range of equipment and schemes and for a wide variety of purposes (U.S. Army Corps of Engineers, 1983; Huston, 1986; Herbich, 1992; Bray et al., 1996; PIANC, 2009; Bray and Cohen, 2010). Most approaches are categorized as either mechanical or hydraulic, with the latter referring to a scheme that involves pumping a water-sediment slurry, often after mechanical loosening of the material being dredged.

A steel I-beam or other device dragged across an underwater high spot can remove a navigation hazard by redistributing sediment underwater and is thus a crude form of mechanical dredging. A clamshell bucket deployed from a standard construction crane on a floating barge can remove submarine sediment and is another example of a mechanical approach. Likewise a backhoe on a barge can function as a dredge in shallow water. Material can be deposited on or in a barge or truck and hauled away for offshore or onshore disposal.

Suction dredges are common for larger projects, and the suction pipe is often equipped with a rotary tool, yielding what is known as a cutterhead suction dredge (Figure 1). The dredge is typically held in place by rigid, vertical spuds, and the cutterhead (Figure 2) lowered to the river- or seabed. The cutterhead can be moved in a sweeping motion across the work area, either by the vessel winching itself or being pushed sideways or by moving the dredge head relative to the vessel (swinging ladder dredge). In this way, the drill head operates a bit like a moving drill bit, biting into the sediment, while a vacuum pump lifts the resulting slurry and pumps it to a barge or neighboring site. Since the material is mechanically mobilized for hydraulic transport, this approach could be defined as a hybrid mechanical/hydraulic scheme.

In many cases, an inline booster pump is used with the hydraulic or hybrid schemes, to overcome head losses within the discharge pipe, allowing discharge at greater distances from the work area. Floating pipe is often utilized to get the slurry to the disposal site. By this approach, dredged materials may be pumped to distances of many kilometers.

Many other types of dredges have been developed. Examples include the horizontal auger dredge, the dustpan dredge, the trailing suction hopper dredge, and the bucket dredge. Suitability of any given design for a particular project depends on the scope of the job, mobilization costs, water depths, sediment characteristics, environmental operating conditions, distance to disposal site, quality and mode of transport of dredged material (spoil), and other factors.

Schemes have also been developed to put sediments into suspension so that naturally occurring water currents will move them away from problem areas. This would obviously increase turbidity significantly, which is often undesirable or prohibited. In other cases, curtains or structures have been installed to reduce the tendency for siltation that would require subsequent dredging.

Material disposal and environmental considerations

The dredged material may simply be disposed of at a convenient site, or it may be moved to a new location where its deposition is considered beneficial, such as for land reclamation or beach nourishment. Offshore disposal is employed in some cases and can often be the least

Dredging, Figure 1 Cutterhead dredge, with spuds deployed at rear, and cutterhead suspended from opposite end.

Dredging, Figure 2 Cutterhead tool lifted clear of the water.

expensive option, but in recent years, more emphasis has been placed on keeping material dredged from coastal areas within the littoral zone, when its characteristics are suitable, to avoid loss of sediments from beaches.

In some cases, the dredged material contains contaminants that must be sequestered. Often this material is placed within an upland confined disposal facility that is dewatered as the material settles (U.S. Army Corps of Engineers, 1987; PIANC, 2002). It can also be placed in a pit underwater and capped (U.S. Army Corps of Engineers, 1998). Vellinga (1997) and Bray (2008) discuss the handling of dredged material containing contaminants. The problem is unfortunately quite common because many of the oldest and largest cities in the world are closely tied to ports and waterways.

Turbidity resulting from dredging activities is often a concern and may restrict available operating times for dredging. Other environmental concerns arise at selected locations and times. In the southeastern United States, for example, dredging is restricted during periods when marine turtles are likely to be in the vicinity of dredging equipment. Many tidal inlets feature shipwrecks that in some instances influence dredging plans or are discovered during dredging.

Summary

Given the large human populations worldwide that reside in coastal areas, and the increasing internationalization and magnitude of commerce, dredging is likely to remain an important global industry. Port capacities will need to continue to be increased, and many ports have the potential to be seriously impacted by relative sea level rise. Dredging schemes will need to be continually improved to increase efficiency and reduce environmental impacts.

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Cross-references

Anthropogenic Impacts
Dredge and Fill
Mass Physical Sediment Properties
Sand Mining/Beach Sand Mining

E

EARTHQUAKE DISTURBANCES

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Definition

Earthquake disturbances result from seismic activity, such as shaking and ground deformation, generated naturally by the rupture of geological faults, or volcanic activity. Human activities such as drilling, mine blasts, and nuclear tests may also simulate seismic disturbances. Direct disturbances generated by earthquakes are usually near field; however, tsunamis may be associated with near- or far-field earthquakes, causing significant effects.

Introduction

One of the earliest reports on the effects of earthquakes on marine communities was from March 20, 1835, when HMS Beagle visited Santa Maria Island, Chile, following the M_w 8–8.5 Concepción earthquake. Darwin (1846) summarized the effects of up to 3 m coseismic uplift, reporting dead and gaping mussels on a nearby rocky flat. Similarly, an earthquake around 400 years ago lifted Kaikoura Peninsula, New Zealand, up to 2 m vertically, stranding the once-productive intertidal lagoons permanently above sea level and altering the food resource available to Māori (McFadgen, 1987). Since earthquakes and their associated tsunami and mass movement events are natural disturbances, there is widespread belief that estuarine communities are largely resilient to their effects over medium to long time scales. In the twenty-first century, however, earthquake events intersecting heavily modified landscapes, such as the March 2011 Great East Japan or Tōhoku M_w 9.0 earthquake and tsunami, have the potential to produce profound future effects in estuaries.

The effects of earthquake events can be especially pronounced when tsunami inundate coastal lowlands, destroying built environments and smothering coastal and fluvial ecosystems with contaminated debris, or when large volumes of pollutants such as fine sediments, trace metals, and wastewater are released by liquefaction and mass movement into estuarine catchments. The 2011 Tōhoku disaster adversely affected hundreds of kilometers of coastal environments, including estuaries, when tsunami waves swept up to 3 km overland and were conducted 3 km farther along river channels (Gomez et al., 2012). In addition to the physical change wrought, damage to the Fukushima Daiichi Nuclear Power Complex resulted in the accidental leak of radionuclides, making fish and shellfish unsafe for human consumption. Initial levels of radionuclides in the hen clam *Pseudocardium sachalinense* reached 950 Ba/kg in Iwaki City, but levels decreased in 2013 to below the regulatory limit of 32 Ba/kg. Similarly high levels were recorded for bottom-dwelling fish, while pelagic fish recorded lower levels (Toyofuku, 2013, "personal communication"). Radioactive contamination has also been detected in the coastal water table and sediments. This disaster has prompted questions regarding the safety of the many nuclear power plants along the tectonically active and tsunami-receiving coasts of the world.

Estuarine coasts are spatially complex environments around which to develop human settlement, often resulting in built environments heavily reliant on bridge systems and lacking inbuilt redundancy in their lifeline networks (transport, water, gas, sewerage, and electricity). Such infrastructure is especially vulnerable to earthquake

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damage due to the effects of liquefaction, lateral spreading, and bank collapse that occur when seismic ground accelerations meet the shallow water tables, coastal and fluvial deposits, and unsupported margins of estuarine environments. As a result of infrastructure damage, untreated sewage, nutrients, and industrial waste may be released into estuaries in large volumes, posing significant health risks to humans, shellfish, and other biota. Following the Canterbury, New Zealand, earthquake sequence of 2010–2011, initiated by the 7.1M_w event in September 2010, estuarine sediment eutrophication and high levels of pathogenic microbes continued for many months (Christchurch City Council data), forcing recreational beach closures of up to 12 months per event (Hart, 2011). The pulses of allochthonous fine sediments disturb the estuary and can contain high levels of contaminants such as trace metals, having negative impacts on estuarine communities (Ilayaraja et al., 2012). Because of increased anthropogenic modification of catchment and coastal areas, sediments, and resources, the effects of earthquakes may become more important in shaping biological communities in the future. Here we review the limited but revealing records of the effects of twentieth- and early twenty-first-century earthquakes and associated disturbances on estuarine communities and environment.

Habitat changes

The ecological effects of large-scale seismic events can be negative or positive and persist for years, decades, and centuries. For estuaries, which are by their nature transitional environments at the interface of fluvial and marine systems, changes can be particularly persistent where significant subsidence or uplift occurs, resulting in changes to their physical (e.g., salinity, exposure) and biological zonation. In Chile, following the 1960 M_w 9.5 Valdivia earthquake and tsunami, land subsidence of more than 1 m in the Rio Cruces resulted in the creation of a large wetland, which is now recognized as a Ramsar site with high biodiversity. After the Chile 2010 M_w 8.8 earthquake, habitat loss and sediment transport have led to the formation of new coastlines with similar geomorphic and habitat properties to those lost. Similarly, following the M_w 9.3 earthquake in the Indian Ocean, recovery of habitats was rapid following the large tsunami of Dec 2004 (Liew et al., 2010). Reports from various parts of the world describe the scouring effects of debris-laden tsunami waters on coastal ecosystems, with sediment removal resulting in the displacement and subsequent removal to higher levels of animals and shells living above mid-tide (Alaska and elsewhere, but not in Canterbury). Where subsidence occurs against heavily modified hinterlands with artificially hardened or recontoured banks, intertidal communities can be lost due to "coastal squeeze." Some of the best pre-quake salt marsh resources of Christchurch City, for example, were lost after the 2010–2012 Canterbury earthquakes when the northern end of the city's estuary subsided in areas where housing and stopbanks had been developed adjacent to the shoreline so that no landward space was available over which the salt marsh could retreat.

Effects on estuarine species

Good Friday Great Alaska M_w 9.2 earthquake of 1964, and subsequent landslides and submarine slumping, which triggered local tsunamis, produced the best documented account of the effects of a seismic event on coastal ecosystems. This account relates to the southern coast of Alaska in Prince William Sound, a 9,000 km semi-enclosed fjord-type estuary that is home to a highly productive, relatively pristine ecosystem supporting extensive populations of seabirds, marine mammals, and fisheries. Vertical displacements of up to 10 m affected the estuary's intertidal mudflats, coastal vegetation, bird nesting sites, and salmon streams (Hanna, 1971; Haven, 1971; Hubbard, 1971; Harwell et al., 2010). On land, "ghost forests" were produced when saltwater inundation and intrusion of the water table killed off and then preserved swathes of coastal pine. On a smaller scale, trees and other terrestrial plants died off, and adjacent salt marsh ecosystems were drowned, along the subsided margins of the Avon-Heathcote Estuary/Ihutai after the Christchurch earthquakes of 2010–2012.

The ecological effects of sudden catastrophic changes in natural habitats have been widely debated with the suggestion that these result in long-term changes, involving several generations of organisms and effects extending over several years (Castilla and Oliva, 1990). The ecological effects of earthquakes on rocky outcrops are thought to be less severe than for soft sediment ecosystems. Mussels, barnacles, and algae are attached organisms normally found on mid- or lower intertidal rocky outcrops. These organisms function as key indicator species and rarely survive coseismic uplift exceeding 2 m due to exposure to air, temperature extremes, and desiccation. Mytilids suffered 100 % mortality following large earthquakes in Alaska in 1964 and Chile in 2010 (Hanna, 1971; Castilla et al., 2010). The 11–60 cm uplift recorded in Chile resulted in initial mortality, shrinking the kelp band, and then the downward vertical extension of kelp beds. The resultant successional changes vacated space, which was invaded by barnacles, enhancing the mosaic areas, increasing diversity, and modifying the shoreline zonation (Castilla and Oliva, 1990). Observations 2 years post-quake recorded a reduction in biomass and no settlement of the dominant mussel.

Following the 1964 earthquake disturbances in Prince William Sound, Alaska, some communities appeared to have recolonized areas within 15 months, with the initial recolonizers differing from pre-quake species in terms of fewer species and individuals. Successional trends were obvious; the pre-earthquake *Verrucaria* zone was colonized first by small filamentous algae and diatoms; and *Fucus*, which had previously dominated the mid-intertidal areas, was replaced by *Porphyra* in the upper littoral and

Ulva sp. in the lower areas (Johansen, 1971). Some 5 years after the quake, however, the original patterns of barnacle distributions had reestablished, *Fucus* had returned to dominance, mussels had reestablished on rocks at lower tidal levels, and the species diversity at MacLeod Island was more diverse than it had been immediately post earthquake.

Generally, the effects on estuarine ecosystems of uplift appear greater than those of subsidence, although relatively little attention has been paid to documenting the latter type of changes. In Prince William Sound, for example, in areas where subsidence had occurred, the overlapping of pre- and post-quake communities was observed, with high intertidal snails found grazing among the submerged terrestrial vegetation.

Wetlands

Coastal wetlands such as salt marshes are vulnerable to earthquake damage and sea-level change since they represent a transition zone between tidal flats and uplands. Due to land subsidence following earthquakes along the Cascadia subduction zone approximately 300 years ago and also in southern Chile (1960), coastal forests were decimated and sedimentation from tsunami activity resulted in the establishment of salt marshes in protected locations with gently sloping shorelines and suitable sediment supplies (FitzGerald et al., 2008). The sudden relative sea-level changes that occur following an earthquake, however, may not mimic models of climate changes where the estimated rates of change are of the order of millimeters to centimeters per year (Reed, 1990). In many areas, the natural recolonization of salt marsh habitats following extreme events has been slow, and intervention may become necessary to restore bird habitats (unpublished data).

In the subtropics, following earthquake and associated disturbances such as tsunami, salt marshes may be replaced by mangroves, which are known to recover rapidly from other natural disturbances such as storms and cyclones. Following the Boxing Day 2004 M_w 9.1–9.3 Sumatra earthquake and subsequent Indian Ocean tsunami, Aceh Province soft-coast mangroves were assessed as being more sensitive to disturbance than rocky headlands, with sandy coasts occupying an intermediate position (Wong, 2009). Past and current records of sea-level changes on soft coasts suggest that mangrove forest resilience, where natural regeneration is expected to take 10–15 years, allows this ecosystem to adapt to both the incremental changes that are associated with global change and the event-scale changes associated with earthquakes and tsunamis (Alongi, 2008). There have been reports of extensive ecosystem damage, with 51–100 % of mangroves destroyed in the Nicobar Islands following the Sumatra 2004 earthquake and tsunami, with similar damage to coral reefs (41–100 %) and to forests (7–27 %) (Ramachandran et al., 2005). This damage, however, is insignificant in comparison to the ecosystem destruction produced in Indonesia over recent decades by clearance for coastal development such as shrimp farms (Idrus, 2009). Reports on the coastal protection value of mangrove forests during the 2004 tsunami event are mixed but, overall, suggest that they provided significant protection to landforms, either through the dissipation of wave energy or through the pre-tsunami progradation of shorelines, in several Indian Ocean countries. While some studies indicate that mangrove forests can provide better storm protection than concrete barriers, the effectiveness will vary depending on location, previous history, and presence of non-mangrove vegetation (Dahdouh-Guebas et al., 2005). Also in the case of a large tsunami or storm surge, coastal forests and dunes are generally believed to have provided minimal hinterland protection, as found during the devastating 2011 Tōhoku earthquake-induced tsunami (Lavigne et al., 2007; Gomez et al., 2010; Gomez, 2012).

Mudflat infauna

Shallow water estuarine mudflat habitats are exposed to multiple environmental stressors, and, according to Thrush et al. (2008), such communities can recover as long as the recovery potential of the resident fauna exceeds the disturbance effects. When this does not occur, then the loss of species leads to habitat loss and fragmentation across landscapes. These types of changes have been observed following earthquake events worldwide.

In shallow water soft sediments, the fauna is often dominated by bivalves which might be expected to have natural resistance to environmental perturbations. In Olsen Bay, Prince William Sound, before the 1964 Alaska earthquake, there were five different species of clams, shellfish resources for people and predators, representing different feeding types; fast and slow burying species; and those that bury to different sediment depths. Mortality patterns after the 1964 Alaskan earthquake were species specific and depended on the degree of sediment displacement and bed uplift, which essentially raised or lowered the animals' vertical distributions. High mortalities were recorded in maximal uplift areas for butter clams (*Saxidomus giganteus*) which, pre quake, were usually buried at depths of 35 cm in the sediment (Baxter, 1971; Hubbard, 1971). In places where the sediment had been removed, shellfish were exposed on the surface and displaced to higher tidal levels where they were unable to bury. For the fast, shallow-burrowing, filter-feeding, little-neck clam *Protothaca staminea*, there was high mortality at upper tidal levels but better survival for larger individuals at lower levels. Post-earthquake surveys suggested that there had been poor reproduction in this species, and also in *Clinocardium nuttallii*, following the quake and also little recruitment. Following the September 2010 and February 2011 earthquakes in Canterbury, New Zealand, another little-neck clam *Austrovenus stutchburyi* demonstrated a high tolerance to earthquake disturbance, with changes in elevation ranging from −0.4 to +0.4 m (Measures et al., 2011). These bivalves maintained their reproductive condition, with recruitment and high survival of juveniles in the year following the earthquakes (own data). Similarly, following the 1964 Alaska earthquake, the deposit-feeding *Macoma inconspicua* appeared to reproduce as usual and settlement occurred. These findings support the hypothesis that following earthquake disturbance, species with large vertical ranges can survive better than those with a more restricted distribution. Recolonization also proceeds faster for more mobile species than their sedentary counterparts, a feature demonstrated by Jaramillo et al. (2012), who investigated the ecological effects of the 2010 Chilean earthquakes on sand beaches.

Seagrass

Many estuarine habitats have seagrass beds at lower tidal levels, and these provide valuable shelter and nursery grounds for fish as well as habitat for mobile invertebrates such as crustaceans. There are numerous reports of damaged seagrass habitats following earthquake disturbances, with suggestions of quite rapid recolonization in some locations. The sensitivity of seagrass to disturbances is well known. Johansen (1971) found that, following earthquake activity, some plants died and the leaves of raised individuals were shorter and narrower than those of the plants remaining at lower tidal levels. In Canterbury, New Zealand, where *Zostera muelleri* has a restricted distribution in anthropogenically affected estuaries, earthquake activity and liquefaction sediment mounds buried up to 70 % of the seagrass habitat during the 2010–2012 earthquakes, with little or no recolonization almost 3 years after the initial damage (own data).

Fish and birds

Because fish are mobile, it might be expected that they can escape the disturbance effects of earthquake activity if alternative feeding grounds and habitats are available. This was, however, not the case for the thousands of rockfish and intertidal fish that died following the 1964 Alaska earthquake (Hanna, 1971). In contrast, the effects of this earthquake on salmonids was reviewed by Noerenberg (1971) and Losey (2005), who summarized the potential effects on chum, and pink salmon, as mortality of eggs and young stages with destruction and siltation of spawning grounds. By tracking the return of the salmonids to individual rivers, it was suggested that less than 10 % of their populations were lost due to the earthquake. For pink salmon returning to subsided streams 3 years after the quake, the decline was close to 8 %, while for uplifted streams the decline was between 40 % and 98 %. Consistent with previous studies, it was concluded that earthquake uplift was more devastating to fishes than subsidence. Chen et al. (2004) looked at fish populations pre and post the 1999 M_w 7.6–7.7 Chi-Chi earthquake and landslide events in Taiwan, reporting considerable annual variations. They concluded that disturbance avoidance behaviors and natural resistance allowed the fish to recover within a few months.

Following the 1964 Alaskan earthquake, scientists also considered the effects on birds and marine mammals: both were considered to be at risk because of their dependence on shoreline-proximal “haul out” areas for resting, breeding, nesting, or feeding. For marine mammals the impacts were uncertain, while the effects on birds may have been reduced because of the timing of the earthquake. Losey (2005) suggested that the most likely impacted groups would be the waterfowl, ducks, and geese, while for herbivorous birds the main impacts would occur as a result of the loss of seagrass habitats. For wading birds the loss of potentially important feeding areas could have an impact on resident and migratory species feeding prior to flying back to their breeding areas. Studies on bar-tailed godwits and oyster catchers in a small estuary in New Zealand impacted by earthquakes in September 2010 and February 2011 have revealed that, despite reduced habitat availability, their invertebrate prey capture rate was similar pre and post earthquake, although this was achieved by using different capture techniques (own data).

Species diversity

Under everyday circumstances, estuarine taxa are able to cope with predictable disturbances including extremes of temperature and salinity and declining oxygen levels. Indeed, it is thought that such disturbances are necessary for maintaining heterogeneity and, therefore, estuarine biodiversity. Whether or not estuarine species diversity is affected by earthquake activity is, thus, a feature of this type of disturbance that is of special interest. In the studies reviewed here, authors have reported a decline in abundance and diversity following earthquakes but then a gradual return to original community characteristics. When a community is stressed, then conservative species generally cope better than those that are opportunistic. Following the 2007 earthquake and tsunami event in Peru, the soft bottom community changed; for example, gammarid amphipods and polychaete worms were absent directly after the tsunami; however, the functional groups and diversity index remained the same. Mechanisms driving this were changes in the sediment including the deposition of finer sediment (Lomovasky et al., 2011). In estuaries with small degrees of seismic uplift, diversity would be expected to be maintained and new habitats rapidly colonized by more mobile species. This would be the case where the sediments are free of contaminants and better draining than the original sediments (own data).

Natural drivers that shape estuarine and other ecosystems include climate, physical/chemical properties, watershed geomorphology, and atmospheric and biological processes (Harwell et al., 2010). In Prince William Sound, the first three are dominant and regarded as being more important than anthropogenic drivers, which include development and resource harvesting. Both anthropogenic disasters, such as large oil spills, and natural events, such

Earthquake Disturbances, Figure 1 Earthquake sediment disturbance (Photo taken on June 15, 2011, from Humphreys Drive Ferrymead, Avon-Heathcote, Estuary/Ihutai, showing the effects of the June 13, 2011, earthquake. Note *pale gray* sands of new sand volcanoes are associated with broader *dark gray* sands of reworked sand from volcanoes produced by February 22, 2011, earthquake. A-A′ approximately 1 m).

as earthquakes and tsunami, can alter habitat such that, on a local scale, coastal habitats become changed. The effects of earthquakes and tsunami, and the associated natural changes in biology, depend on several major factors, including the degree of land and bed elevation changes, relations to recruitment seasons, the life history characteristics of species, and the degree of interaction with human-altered environments which can act as a source of contamination and debris and provide fixed boundaries that limit subsequent, post-quake coastal and estuarine adjustments.

Geological and geomorphic effects

Two of the most apparent and immediate effects of earthquakes on estuary geomorphology concern the disturbance of bed sediments and levels, with immediate flow on effects on tidal inundation patterns and longer-term effects on estuary hydrology, shoreline stability, and ecological zones. The saturated, soft sediments of estuaries are prone to liquefaction during shaking produced in earthquakes, expressed as the surface rupture of sand volcanoes, as well as sediment compaction and subsidence. Bed levels are also affected by underlying bedrock deformation, although the surface expression of uplifted bedrock may be dampened by the compaction of overlying sediments. During the February 2011 Christchurch, New Zealand, M_w 6.3 earthquake, up to 0.5 m uplift and subsidence were produced in the southern two thirds and northern third of the Avon-Heathcote Estuary/Ihutai, respectively, reducing the overall tidal prism by 14 % or one million liters and the mid-tide wetted area by 18 % (Measures et al., 2011). These changes occurred due to both soft sediment settling and bedrock deformation, with subsequent changes in the estuary's entrance channel and shoreline expected in the future as the hydraulics adjust to the altered bed. Over 40 % of the surface of this estuary's bed was covered in sand volcanoes (photo, Figure 1).

Vertical changes in estuaries shift intertidal zones and move organisms out of their preferred tidal exposure range. Mobile organisms may relocate quickly if suitable habitat is available, while fixed plants and animals migrate over several reproductive cycles and seasons where space is available. Severe shaking can also disconnect tree root systems from the mycorrhizal fungi allowing their nutrient intake, killing the vegetation over periods of several years. Following the February 2011 earthquake in Christchurch, plants with very specific high intertidal preferences, such as the glasswort *Sarcocornia*, were observed, the seedlings of which shifted noticeably landward or seaward within a year, depending on the subsidence or uplift of their habitat, with the displaced settlement of new seedlings becoming more pronounced over the subsequent 2 years. The landward migration of this vegetation was, however, hindered along the stopbanked northern margins of the estuary, where considerable salt marsh habitat was lost due to "coastal squeeze": the horizontal reduction of habitat space that can occur where relative water levels rise adjacent to engineered waterbody margins.

In terms of the geological record, microorganisms such as foraminifera play an important role since these indicator species are readily preserved, providing evidence of prehistoric earthquake changes. Estuarine foraminifera are sensitive to salinity and tidal exposure (Hayward et al., 1999a). Specific taxa indicate salt marsh environments and have been used to track gradual Holocene sea-level changes (Hayward et al., 1999b; Gehrels, 2000). They can also record sudden changes such as earthquake-induced bed level changes and, along with diatoms and other microorganisms, have been sampled from estuarine cores to determine the recurrence intervals of seismic events (Cochran et al., 2007; Hayward et al., 2010). Analysis of floral and faunal data collected from cores, combined with sediment dating, can indicate both the scale and recurrence intervals of seismic changes in estuary beds. This provides vital background information for planning development around estuaries since building foundations and lifeline networks above- and below-ground, including roads, bridges, water, gas, and sewerage systems, are very vulnerable to earthquake-induced soft sediment disturbances. Catastrophic lifeline network failures that have resulted from the disturbance of soft sediments around coastal and estuarine margins include the destructive effects of the 1886 M_L Charleston earthquake, the 2011 Tōhoku earthquake, and the 2010–2012 Canterbury, New Zealand, earthquake sequence.

Soft sediment disturbances can also provide a geological record of seismic events in estuaries. During the 2010–2012 Canterbury, New Zealand, earthquake sequence, each seismic event that exceeded 5.2 M_W produced extensive sand volcanoes (Reid et al., 2012; Quigley et al., 2013). At the surface, the sand volcanoes were quickly modified by tide and winds, resulting in broad sandy hummocks, whereas in the subsurface, small but distinctive pipes were produced that remain preserved despite the surface reworking of the estuary bed (Reid et al., 2012). The preserved pipe structures of sand volcanoes may be associated with other types of soft sediment folding and distortion (see Montenat et al., 2007).

Seismic events, either in nearby coastal regions or far field, may also produce tsunami depending on the earthquake magnitude and the geometry of fault rupture and seafloor change. Tsunami may be recorded as distinct horizons within estuarine sediments, with signature marine micro- and macrobiota and outsized clasts transported by the tsunami and left relict and unable to be remobilized by estuarine hydrodynamics. The ongoing influence of the influx of marine water can include elevated salinity levels and the physical removal of surface plants and organisms.

Summary

There is no dispute that earthquakes and their associated tsunamis and mass movements affect estuarine ecosystems. The effects are often dramatic. While some habitats are lost, new habitats may be made available. The recovery of estuarine communities following an extreme earthquake disturbance will generally follow a successional sequence according to the physical and chemical conditions in the environment, the local hydrodynamic conditions, and the availability of potential recruits.

Earthquakes and their associated tsunami and mass movements are natural events, and when human environment interactions are minor, recovery of most estuarine systems is predicted to occur within 4–10 years (Borja et al., 2010), probably more quickly than in the slower-growing habitats of tropical reefs (Phongsuwan and Brown, 2007). This was the case, for example, for the estuarine habitats affected by the 1964 Alaskan earthquake and tsunami and for the 1960 Chilean events (Reinhardt et al., 2010). It is, however, not the case for the earthquake, tsunami, and subsequent nuclear disaster that occurred along the Tōhoku coast of eastern Japan in 2011, including some of the most heavily modified and populated coastal landscapes of the early twenty-first century (Gomez et al., 2012). Given current global trends toward the coastal concentration of human populations, industry and infrastructure including seawalls and nuclear power plants, and the associated modification of catchments, and coastal plains and margins, the effects on estuaries of earthquakes and their associated natural and human-induced events are likely to be more pronounced and endure within the landscape for decades to centuries (Hart, 2011).

Some scientists suggest that recovery from natural disturbances will be faster than those caused by anthropogenic changes. Determining the time scale of recovery from earthquake disturbances continues to be a challenge, with reported estimates varying from days for some species to years and decades and periods up to 25 years. Long-term studies have been unable to quantify this. Moreover, the challenge is likely to grow as earthquake, tsunami, and mass movement events interact with increasingly human modifications to coastal and linked environments worldwide over the twenty-first century, mixing the natural and human-produced effects of these events. It is clear, however, that from the evolution of estuarine habitats, the new restored habitat may be different from that originally damaged by earthquake activity. One important feature of this is whether this is a natural result or one produced by human activity restricting the ability of an estuary to recover. Nevertheless it would be expected that the mosaic of communities formed after the disturbance would maintain species diversity.

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Cross-references

Fish Assemblages
Infauna
Macrofauna
Mangroves
Sediment Transport
Soft Sediment Communities

ECOLOGICAL MODELING

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Synonyms

Ecosystem modeling

Definition

Ecological modeling (noun): The field of study in which biological (i.e., biotic) and environmental (i.e., abiotic) processes are represented mathematically to enable quantitative analysis and simulation of individuals, populations, communities, and ecosystems.

See Box 1 for related definitions.

Introduction

> A complex field such as oceanography tends to be subject to two opposite approaches. The first is the descriptive, in which several quantities are measured simultaneously and their inter-relationships derived by some sort of statistical method. The other approach is the synthetic one, in which a few reasonable although perhaps oversimplified assumptions are laid down, these serving as a basis for mathematical derivation of relationships. Gordon A. Riley (1946)

With this introduction, Gordon Riley embarked on his description of the first mechanistic numerical model in a marine ecosystem, in this case a simulation of phytoplankton biomass on Georges Bank. This classic work illustrated the power of the synthetic approach in marine science and laid the groundwork upon which the field of estuarine ecosystem modeling was later built. Riley's work provided a significant development upon the Lotka-Volterra equations of predator-prey dynamics (Lotka, 1925, 1932; Volterra, 1926):

$$\frac{dN_1}{dt} = N_1(r - gN_2) \tag{1}$$

$$\frac{dN_2}{dt} = N_2(fgN_1 - d) \tag{2}$$

where N_1 and N_2 are the size of the prey and predator populations, respectively, r is the intrinsic growth rate of the prey, g is the grazing or attack rate of the predators, f is the efficiency with which predators translate consumed food into new offspring, and d is the death rate of the predators. The key advancement provided by Riley was to formulate these processes as functions of environmental variables such as temperature, irradiance, and nutrient concentration. By specifying measured values for these "forcing functions," Riley was able to predict annual plankton cycles which matched the observations remarkably well. Riley's models of phytoplankton (Riley, 1946) and zooplankton (Riley, 1947) on Georges Bank were subsequently combined into the first coupled nutrient-phytoplankton-zooplankton (NPZ) model for the western North Atlantic (Riley et al., 1949).

Riley's mechanistic approach formed the basis for the field of marine and estuarine simulation modeling which developed in earnest in the 1960s and 1970s (Figure 1 and references therein; Wetzel and Wiegert, 1983; Hopkinson et al., 1988; Hofmann, 2000; Brush and Harris, 2010). Riley's approach to mechanistic simulation of biological compartments (e.g., phytoplankton, zooplankton) was combined with the similar mechanistic approach to simulating abiotic state variables such as the Streeter-Phelps dissolved oxygen model (Streeter and Phelps, 1925) to develop increasingly complete ecosystem models. These early models were used primarily for heuristic understanding of ecosystem structure and function and were gradually expanded from the original NPZ structure to include multiple primary producers (e.g., phytoplankton, sea grass, benthic algae) and

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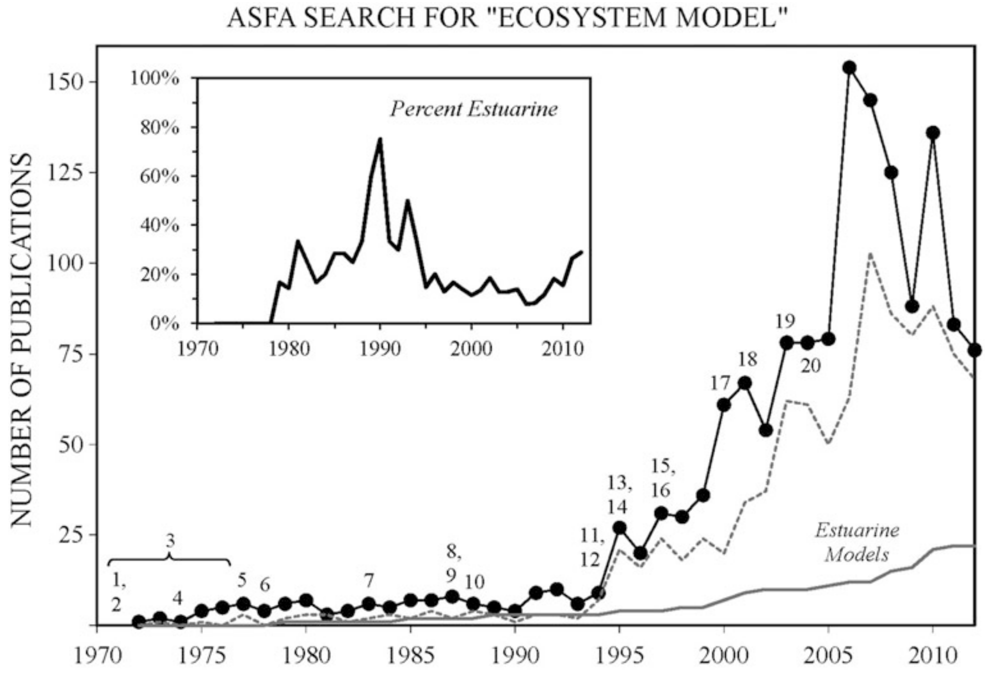


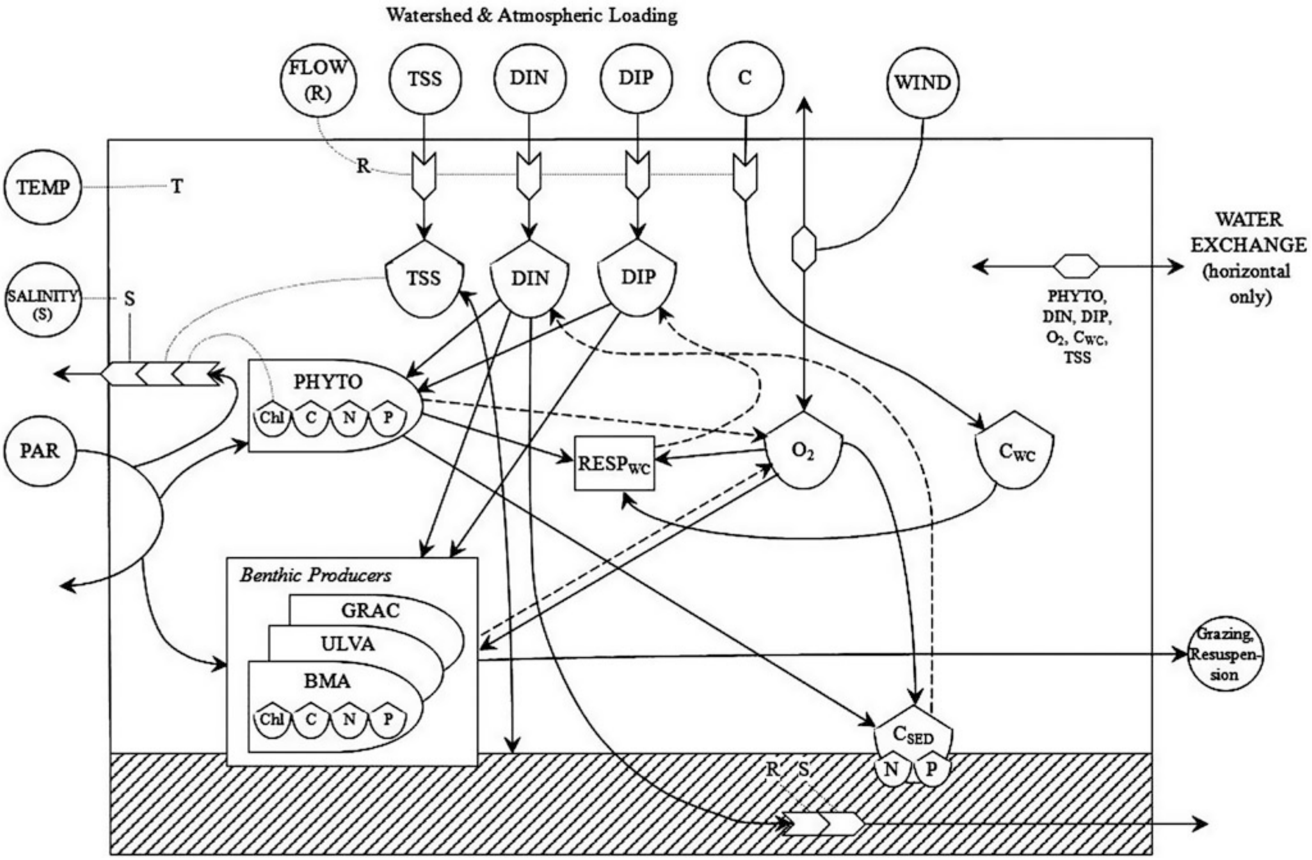
Ecological Modeling, Figure 1 Number of publications returned using the term "ecosystem model" in the Aquatic Sciences and Fisheries Abstracts (ASFA) online database, with some major milestones in coastal marine and general ecosystem modeling highlighted with numbers: *1*, DiToro et al. (1971); *2*, Odum (1971); *3*, Patten (1971, 1972, 1975, 1976); *4*, Steele (1974); *5*, Hall and Day (1977); *6*, Kremer and Nixon (1978); *7*, Odum (1983); *8*, Thomann and Mueller (1987); *9*, HydroQual (1987); *10*, Baretta and Ruardij (1988); *11*, Odum (1994); *12*, Cerco and Cole (1994); *13*, Rigler and Peters (1995); *14*, Baretta-Bekker (1995); *15*, Chapra (1997); *16*, Baretta-Bekker and Baretta (1997); *17*, Odum and Odum (2000); *18*, DiToro (2001); *19*, Canham et al. (2003); and *20*, Cerco and Noel (2004). Broken *grey line* shows the number of publications in scholarly journals only. Solid *grey line* shows the subset of all publications also containing the terms "estuary," "bay," or "lagoon" (but not "lake") as an indication of the fraction of models from estuarine and similar ecosystems; inset expresses this result as a percent of the total (Updated from Brush and Harris (2010)).

functional groups (e.g., diatoms and dinoflagellates, mesozooplankton, and microzooplankton), biogeochemical cycling of multiple carbon and nutrient pools, dissolved oxygen, suspended sediments, and multiple consumers including mesozooplankton and microzooplankton, benthic infauna, and fish (e.g., Figure 2).

These heuristic models began to be used to address management questions in the 1980s and 1990s, particularly related to the effect of anthropogenic nutrient enrichment on estuarine eutrophication (e.g., HydroQual, 1987; HydroQual, 1991; Cerco and Cole, 1994; HydroQual and Normandeau Associates, 1995). The use of models practically exploded in the late 1990s (Figure 1) due to the widespread acceptance of models as mainstream research tools and the increasing availability of personal computers capable of running simulation models (Canham et al., 2003; Solidoro et al., 2009; Brush and Harris, 2010), although this increase may be somewhat overemphasized in the primary literature since many early models were published as book chapters or in the grey literature. Regardless, the use of models in both research and management has continued to grow, and models have become central to efforts to manage nutrient loading and mitigate cultural eutrophication in coastal systems (US EPA, 1999; NRC, 2000; Giblin and Vallino, 2003; Harris et al., 2003; US EPA, 2010).

Why model?

As noted above, models first emerged as heuristic tools to enhance our understanding of estuarine structure and function. Haefner (2005) describes models as hypotheses about how a system works. They provide tools by which we analyze, synthesize, and test our understanding of systems through retrospective and predictive scenarios (Fennel and Neumann, 2004). Models provide a way of dealing with the inherent complexity, variability, and open nature of aquatic systems, which can make them difficult to study using traditional field and experimental methods. They provide a means for scaling up typically sparse measurements in both space and time and quantifying processes which have not been measured (or which may be unmeasurable). Given these capabilities, models have become fundamental components of interdisciplinary research programs, providing a powerful means of synthesizing our understanding, integrating diverse datasets, and highlighting gaps in our knowledge (Kemp and Boynton, 2012).



Ecological Modeling, Figure 2 Sample model diagram using the energy systems language of Odum (1983, 1994). *BMA* benthic microalgae, *C* carbon, *Chl* chlorophyll-a, *CSED* sediment carbon, *CWC* water column carbon, *DIN* dissolved inorganic nitrogen, *DIP* dissolved inorganic phosphorus, *FLOW* freshwater inflow, *GRAC Gracilaria tikvahiae*, *N* nitrogen, *O_2* dissolved oxygen, *P* phosphorus, *PAR* photosynthetically active radiation, *PHYTO* phytoplankton, *RESPWC* water column respiration, *TEMP* water temperature, *TSS* total suspended solids, *ULVA Ulva lactuca*, *WIND* wind speed.

Finally, models provide a means for conducting "whole-system experiments" which are nearly impossible to do outside the virtual realm in coastal systems. Some of our most fundamental understanding of aquatic systems has come from relatively rare, real-world, whole-system experiments which capture the response of entire ecosystems. Arguably the prime examples of these are the lake fertilization experiments of Schindler (1974) in the Experimental Lakes Area of Canada, which cemented our understanding of phosphorus as the primary limiting nutrient in temperate lakes and the cause of lake eutrophication in the 1960s and 1970s. While some early examples of whole-system experiments exist for estuaries, their open connection to the sea and increasing governmental protections make these sorts of whole-system experiments nearly impossible outside of mesocosms (Nixon et al., 1986). Models provide a way of conducting virtual whole-system experiments not possible in the real world. One can ask how an estuary will respond to changes in nutrient loading, climate, fishing pressure, food web structure, or restoration. This "what if?" capability gives models a predictive, forecasting capability which forms the basis of their use in informing estuarine management.

Below we summarize the major uses of models in estuarine science:

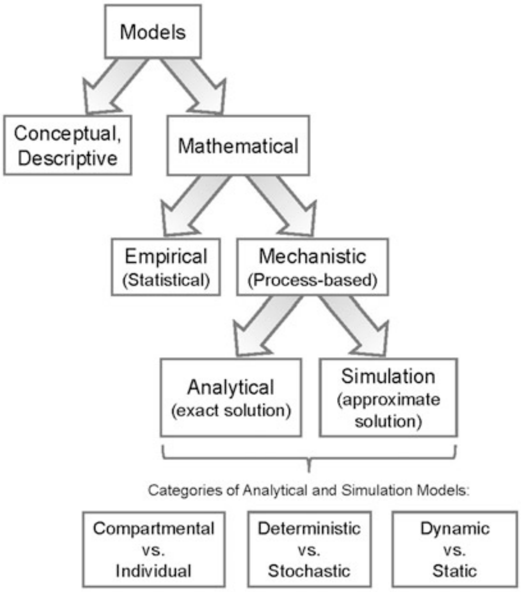
1. *Heuristic* (research) tool – understanding system structure and function, hypothesis testing, estimating values in the absence of observations, and quantitative explanation of ecological theories and empirical models
2. *Synthesis* of extensive datasets – scaling data over time and space (i.e., interpolation and extrapolation), creating system-level budgets, identifying gaps in knowledge, and guiding sampling methods and monitoring programs
3. *Simulation analysis* – ability to conduct whole-system "what-if?" experiments and prediction/forecasting of future ecosystem states
4. *Management* – simulation of estuarine response to nutrient management, setting total maximum daily loads (TMDLs) of pollutants, and development of restoration plans

Types of models

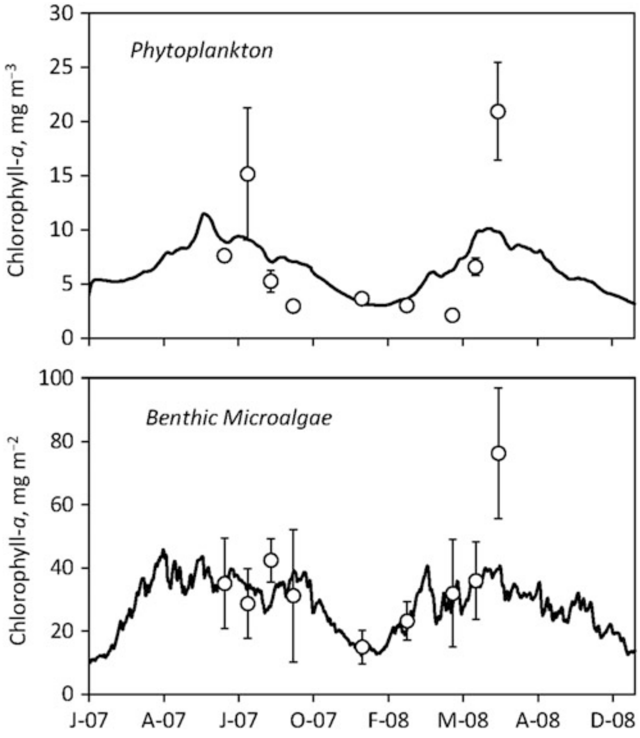
At the most basic level, models can be divided into conceptual and mathematical models (Figure 3). The former present a diagrammatic synthesis of the interrelationships between components of an estuary and allow one to draw qualitative predictions about how a system will respond to perturbation. Mathematical models do the same, except that the relationships are formulated with equations to allow quantitative prediction. Mathematical models can be divided into empirical and mechanistic models; the former are statistical relationships between variables which can often provide powerful predictive capability. Classic examples for aquatic systems include the phosphorus-chlorophyll models of Dillon and Rigler (1974) and Vollenweider (1976) for lakes, which have been extended into estuaries using nitrogen as the limiting nutrient and chlorophyll, primary production, and fisheries yields as the response variables (e.g., Nixon, 1992; Boynton and Kemp, 2000; Nixon et al., 2001). However, statistical models do not provide any underlying explanation of the processes involved (i.e., "correlation does not imply causation").

Mechanistic or process models attempt to provide this explanation by developing equations that piece together individual physiological, ecological, and behavioral formulations that relate ecological rates (e.g., growth, consumption, respiration) to environmental or biological factors (e.g., biomass, temperature, light, nutrients). Some mechanistic models are simple enough that exact analytical solutions can be determined through integration (i.e., an exact estimate of biomass or concentration at any future time). However, most are too complex for an exact analytical solution and must be solved over successive time steps in a process called numerical iteration (i.e., simulation models). The result is predicted biomass or concentration through time (Figure 4).

Mathematical models of estuaries can be further subdivided in numerous ways. We summarize a few major categories here (see also Figure 3 and Box 1). Models designed to simulate trophic interactions in estuarine and coastal systems can be divided between those taking a population approach (simulating numbers of organisms using variations of the Lotka-Volterra equations) and a systems ecology approach (simulating compartmental biomass or concentrations using mechanistic formulations) that includes limited details on community structure. The former is typically applied to single species of higher-trophic-level (HTL) organisms, while the latter is more typically applied to lower-trophic-level (LTL) processes. This dichotomy reflects the difficulty of producing models that can simulate both water quality and realistic population dynamics of HTLs. Indeed, development of end-to-end (E2E) models capable of simulating dynamics from nutrients through fish is a major challenge (and currently a major area of research) given the very different time scales of key rate processes and increasing complexity of life histories, life cycle processes, and importance of migration as one moves up the food chain.



Ecological Modeling, Figure 3 A classification scheme for models. See text and Box 1 for details and definitions.



Ecological Modeling, Figure 4 Sample model output (*lines*) compared to observations (points ± standard error) for phytoplankton and benthic microalgal biomass in Hog Island Bay, VA (Source: Brush and Harris, unpublished).

Mechanistic formulations in estuarine ecosystem models have largely focused on lower trophic levels and water quality (e.g., NPZ models), especially the relationships between nutrient loading and light and how these factors impact phytoplankton growth, survival of submerged grasses, and development of hypoxia/anoxia (e.g., Kremer and Nixon, 1978; Cerco and Noel, 2004). In contrast, models focused on higher trophic levels, such as Ecopath with Ecosim (EwE, see www.ecopath.org), tend to contain fewer mechanistic functions. Both of these approaches make attempts at "ecosystem" conditions in somewhat simplified terms; LTL models represent the higher trophic levels as a simple closure term, and HTL models represent the lower trophic levels as a forcing term. In fisheries management applications, modeling efforts have typically employed multiple species approaches such as EwE, MSVPA (multispecies virtual population analysis) models, or multispecies production models, while more mechanistically formulated examples typically take the form of individual-based or bioenergetic models (Latour et al., 2003; Travers et al., 2007). An intermediate step toward development of E2E models involves coupling of single-species models of shellfish and fish to larger, compartmental LTL models (e.g., Cerco and Noel, 2007).

Network analysis provides another means for predicting the exchange of energy and materials amongst various ecosystem compartments and through entire food webs using mass-balance constraints (Dame and Christian, 2006). As a precursor to the now ubiquitous EwE models, network analyses such as those of Baird and Ulanowicz (1989) and Ulanowicz and Wulff (1991) have been used to generate metrics of energy flow that could be used for estuarine comparison. Applying network analyses in the Chesapeake and Baltic, Ulanowicz and Wulff (1991) were able to quantify characteristics related to the transfer and flow of energy through the two ecosystems to make so-called anatomical comparisons in the structure and composition of ecosystem components. While these network analyses can be very effective at revealing ecosystem structure, they are data intensive and may not provide the type of mechanistic understanding or forecasting capability frequently desired for management applications.

Nutrient and energy budget and mass-balance approaches are another modeling approach that has been especially useful in estuarine comparisons and as tools for management (Boynton and Nixon, 2012). These types of models are based on the principles of conservation of mass and energy, where the goal is to quantify all input and output terms (e.g., carbon and/or nitrogen) for a particular marine system. Considerable insight can be gained by comparing these budgets for different marine systems even when there are unquantified terms (see examples in Kremer et al., 2000). While these mass-balance models are not typically dynamic, they provide a numerical framework for understanding major processes and inputs to a given ecosystem.

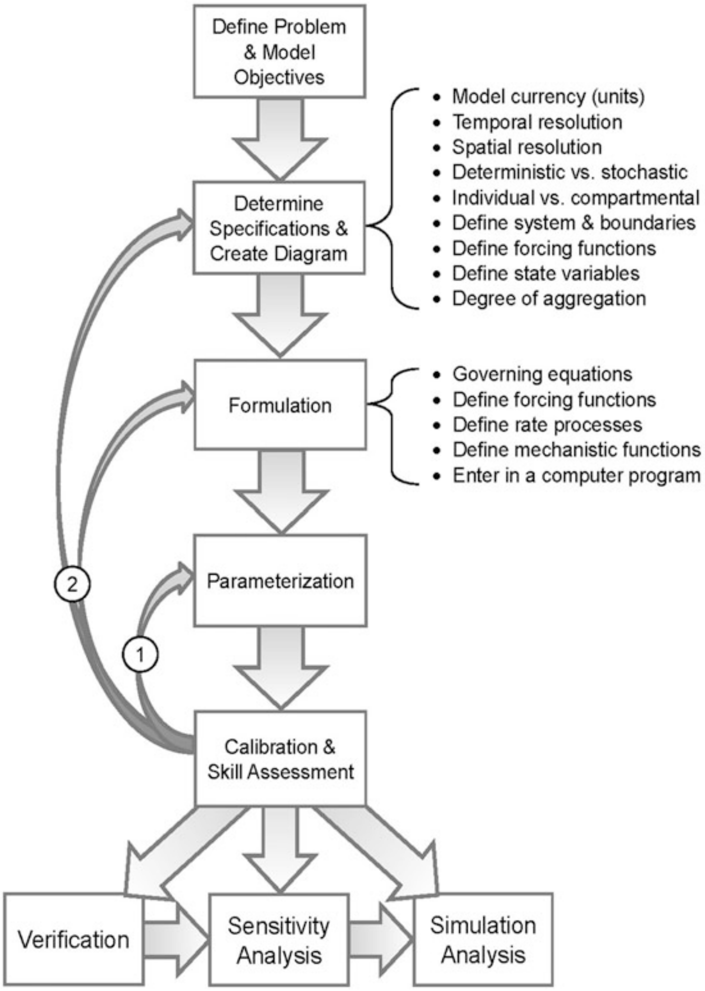
The modeling process

Models are developed through a process that proceeds from conception through formulation, testing, and application (Figure 5). The type and complexity of the model used are dependent upon the research question or application. The modeler must first determine the boundary of their system to be modeled, both in terms of space (i.e., the model domain) and what to include (i.e., forcing functions vs. state variables). The modeler must then decide how much biological and biogeochemical complexity to include, balancing the need for a realistic model with data available for formulation and model testing (i.e., calibration and verification). Similarly, the modeler must decide on the appropriate spatial resolution, from representing the estuary as a single box, through one- and two-dimensional arrays of boxes, to a highly resolved 3D mesh. These choices determine the balance between model precision (i.e., ability to reproduce observations with minimal error), realism (i.e., degree to which the model is an accurate representation of the real system), and generality (i.e., degree to which the model is readily transferable to new systems) (Levins, 1966). Recent cases of "participatory modeling" have provided examples of how the numerical modeler may communicate and interact with non-modeling end users to make these choices and have provided early successes in how this approach may improve the applicability of model output to real-world problems (Voinov and Bousquet, 2010).

Once these choices are made, the model formulations are developed and parameterized. The completed model must then be tested against (i.e., calibrated to) a set of observations to maximize the fit between model and data as much as possible. At this point, modeling becomes an iterative process, in which one will need to go back in the process to adjust parameter values within reasonable ranges to improve the fit. If the fit does not improve, one may need to go further back to reconsider some of the formulations that were chosen or even further back to reevaluate the structure and assumptions of the model. Once calibration is successful, the model is then ideally verified (also termed "validation" or "confirmation") against an independent dataset – perhaps a portion of the observations held back from calibration or data from a different system or time period. The goal of verification is to reproduce the independent dataset without needing to make further changes to parameter values or model structure. Once one has confidence in the quality of the model, its behavior can be further analyzed via sensitivity analysis and skill assessment, or the model can be used to address the original research question via heuristic exploration, hypothesis testing, scenario analysis, and/or management application.

Summary

Ecological and ecosystem models have emerged as powerful research, synthesis, and management tools over the



Ecological Modeling, Figure 5 The modeling process. See text and Box 1 for details and definitions. Feedback 1 represents adjustment of parameters to improve calibration, which may or may not be automated. Feedback 2 represents more fundamental changes in model structure and formulations that may be required if 1 is unsuccessful.

last several decades. They serve as important tools for synthesizing current understanding and available data, understanding how estuaries function, testing our assumptions, and identifying gaps in our knowledge. They provide us a means of conducting whole-system experiments to identify how estuaries respond to change, and this predictive capability has made them essential tools for informing estuarine management. A wide variety of modeling approaches exist from conceptual to mathematical, individual to compartmental, and population to ecosystem. Models will continue to play a critical role in both research and management in estuarine systems and can be continually updated as our empirical understanding of ecosystem processes improves. In addition to their ongoing role in predicting estuarine response to changes in nutrient loading, fishing pressure, and restoration, models will become increasingly important in the coming years in predicting responses to climate change, individually and in combination with other stressors.

Box 1: Modeling definitions

Analytical model–A model with equation(s) that can be mathematically solved to produce an exact solution or prediction at any time.

Boundary conditions–The values of state variables imposed at the boundary of the model domain; these are required when water mixes into the model domain from outside the system.

Calibration–The process by which parameter values are modified within reasonable ranges by the user, or with an optimization routine, to obtain the best possible fit of model predictions to observed data. Calibration is sometimes referred to as “fitting” or “tuning” a model.

Compartment model–A model that simulates pools of biomass, concentration, numbers, or energy content aggregated across populations or communities rather than of individuals.

Conceptual (or descriptive) model–A diagrammatic representation of the feedbacks and relationships operating within a system; these models synthesize existing understanding and enable one to make qualitative predictions as well as decisions regarding numerical model formulation.

Deterministic model–A model that uses exact values for all parameters (no uncertainty) and produces a single prediction for each state variable at each point in time.

Domain–The spatial extent of the model, i.e., the area or volume being modeled.

Dynamic model–A model that produces predictions that vary over time.

Empirical model–A model based on statistical relationships between variables of interest rather than theoretical or mechanistic relationships.

Forcing functions–Exogenous variables that impact the system but are not explicitly modeled; instead, they are input or “forced” into the model.

Formulation–(Verb) The process by which model equations are developed (as in Figure 3) or (noun) a model equation.

Individual-based (or agent-based) model (IBM)–A model that simulates the unique behavior of individuals (or agents) in a population; IBMs apply the perspective that ecosystem patterns emerge as a product of the interactions of individuals with each other and their environment.

Initial conditions–The starting values used for each state variable at the beginning of a model run.

Mechanistic model–A model that formulates processes as the combined result of physiological, biogeochemical, physical, and/or behavioral mechanisms which are typically functions of environmental or model state variables.

Model–A conceptual or mathematical simplification (or abstraction) of a real system.

Model currency–The units used to simulate state variables and rate processes in the model (e.g., $\mathrm{g\,C\,m^{-3}}$, $\mathrm{g\,C\,m^{-3}\,d^{-1}}$).

Optimization–An automated process in which parameters are adjusted to achieve the best possible fit to observations.

Parameterization–The process in which values or distributions are chosen for model parameters, typically from the literature, previous models, and field and/or experimental data.

Parameters–Components (e.g., intercepts, slopes) of both empirical and mechanistic model formulations which can be fixed (deterministic models) or allowed to vary (stochastic models).

Sensitivity analysis–The process by which individual parameters of a model are changed by some amount (e.g., ± 10 % or within known ranges) and the effect on model predictions (e.g., % change in predicted state variables) is quantified. Sensitivity analysis can also be performed on initial conditions, boundary conditions, and forcing functions (although modification of forcing functions is typically referred to as simulation analysis).

Simulation (or numerical) model–A model with equation(s) that do not have an exact solution and therefore must be solved over successive time steps, a process called numerical iteration.

Simulation (or scenario) analysis–The use of a model to understand system structure and function; this often includes “what if?” scenarios in which the model is used to predict system response to changes in an important parameter or forcing function.

Skill assessment–Statistically quantifying the degree to which the model reproduces the data (i.e., the model-data misfit).

Spatial resolution–The degree to which space is resolved by a model; models are typically 0D (single point or box), 1D (e.g., boxes along the axis of a system or layers through the water column), 2D (e.g., boxes along the axis of a system with >1 depth layer), or 3D (e.g., gridded models that resolve the x, y, and z dimensions).

Standard run–The final, best-fitting predictions of a model given the data available for calibration which forms the basis for skill assessment, sensitivity analysis, and simulation analysis.

State variables–Quantities within the system to be modeled through time (e.g., biomass, concentration, numbers, energy content).

Static model–A model that produces predictions that are constant over time; however, these models can be solved iteratively making them quasi-dynamic.

Stochastic model–A model that incorporates uncertainty in parameter values and produces predictions with error distributions or a set of possible model trajectories.

Temporal resolution–The unit of time for which rate processes are resolved by a model.

Time step–The increment of time over which successive iterations of a model are solved.

Verification (also called validation or confirmation)–The process by which a model is tested against an independent dataset (e.g., different year or system) without changing the parameter values used in calibration.

Box 2: How modeling works: the biologist's toolbox

Here, we provide an example of how ecological models work by focusing on a time-dynamic, mechanistic example. Suppose we want to predict the biomass of phytoplankton, P (g C m^{-2}), over an annual cycle. Changes in P over time are due to differences in the input of new biomass (i.e., primary production or growth) and the output of biomass through various loss processes. If Phytoplankton grow under non-limiting conditions (e.g., in culture), they will increase exponentially, and we can develop an expression for the instantaneous rate of increase at any point in time:

$$\frac{dP}{dt} = k_1 P \tag{3}$$

where k_1 is the instantaneous or intrinsic rate of increase in units of per time (e.g., d^{-1}). Similarly, the population will exponentially decrease in the absence of light and nutrients as respiration or mortality (e.g., k_2) are the dominant processes:

$$\frac{dP}{dt} = -k_2 P \tag{4}$$

We can combine these equations to allow both processes to occur simultaneously:

$$\frac{dP}{dt} = k_1 P - k_2 P \tag{5}$$

This "governing equation" for phytoplankton biomass can have any number of inputs and losses. Ignoring advection and diffusion, we might have growth balanced by respiration (k_2), sinking (k_3), and grazing (k_4):

$$\frac{dP}{dt} = P(k_1 - k_2 - k_3 - k_4) \tag{6}$$

This equation gives us the instantaneous rate of change of the population, but what we really want to predict is biomass through time. We can convert the differential equation into the discrete, finite difference form:

$$P_{t+1} = P_t + P_t(k_1 - k_2 - k_3 - k_4)\Delta t \tag{7}$$

where P_t and P_{t+1} are biomass at two successive points in time, and Δt is the interval over which we are solving the equation, in this case 1 day. Suppose $P_0 = 1$ g C m^{-2} and $k_1 = 0.7$, $k_2 = 0.2$, $k_3 = 0.1$, and $k_4 = 0.1$ d^{-1}. Biomass after 1 day is then

$$\begin{aligned} P_1 &= 1 + 1\cdot(0.7 - 0.2 - 0.1 - 0.1)\cdot 1 \\ &= 1.3 \ \ \mathrm{g\,C\,m^{-2}} \end{aligned} \tag{8}$$

We use the computed value after the first day as input to compute biomass after the second day:

$$\begin{aligned} P_2 &= 1.3 + 1.3\cdot(0.7 - 0.2 - 0.1 - 0.1)\cdot 1 \\ &= 1.69 \ \ \mathrm{g\,C\,m^{-2}} \end{aligned} \tag{9}$$

This can then be repeated for as many time steps as desired. This numerical iteration forms the basis for how time-dynamic estuarine models compute biomass or concentrations through time. However, the rates are not constant in real systems, but rather change as a function of environmental or biological variables. For example, phytoplankton growth is generally considered to be a function of water temperature (T), irradiance (I), and nutrient concentration (N) (e.g., Kremer and Nixon, 1978). Changing notation from k_1 to G for growth rate, models often set the maximum daily growth rate ($G_{\max}$) as an exponential function of temperature, for example, by using Eppley's (1972) classic formulation:

$$G_{\max} = 0.59e^{0.0633T} \tag{10}$$

This maximum rate must be reduced to account for suboptimal (i.e., limiting) irradiance and nutrients. Growth rate is typically a saturating function of both factors:

$$G = G_{\max}\left(\frac{I}{k_I + I}\right) \tag{11}$$

$$G = G_{\max}\left(\frac{N}{k_N + N}\right) \tag{12}$$

where k_I and k_N are the "half-saturation" constants or the values of I and N at which growth is half the maximum rate (there are numerous ways to formulate these relationships but these are the simplest). Normalizing both functions to the maximum rate,

$$LTLIM = \frac{G}{G_{\max}} = \frac{I}{k_I + I} \tag{13}$$

$$NUTLIM = \frac{G}{G_{\max}} = \frac{N}{k_N + N} \tag{14}$$

produces dimensionless fractions from 0 to 1 which can be multiplied by the $G_{\max}$ function to reduce growth in the case of limiting conditions:

$$G = G_{\max}\cdot LTLIM\cdot NUTLIM \tag{15}$$

Time-dynamic mechanistic models piece together these types of formulations for each rate in the model as appropriate, with typical relationships being formulated as linear, exponential, saturating, and power functions (Haefner, 2005). For example, respiration and nutrient recycling terms are typically exponential with temperature. Zooplankton growth is typically formulated as a maximum rate exponentially dependent on temperature, multiplied by a dimensionless, saturating function of phytoplankton biomass (e.g., *FOODLIM*). Growth of larger animals such as oysters, clams, and fish is typically formulated as an allometric function of individual weight, followed by scaling terms for temperature and food. Often, terms in the governing equations are linked to other variables being modeled, e.g., nutrients in the *NUTLIM* equation or zooplankton biomass in the grazing rate formulation.

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Cross-references

Ecosystem-Based Management
Seagrass Production Models

ECOLOGICAL MONITORING

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Definition

Monitoring is quantitative sampling through time to detect and measure changes in chemical, physical, or ecological variables, especially related to environmental impacts.

Introduction

Monitoring is a frequent activity done for various purposes, often the detection and management of environmental disturbances (fishing, pollution, dredging, etc.), including long-term changes, such as the predicted effects of climatic change. Monitoring is routine sampling at specified times, intervals and places to determine starting conditions and subsequent changes in ecological measures. These include univariate measures (e.g., abundance of a single species) or multivariate measures (e.g., patterns of concentrations of a suite of hydrocarbons). Whatever the measure, the objective is to detect changes in what is being measured and to identify (where possible) whether the changes are natural or man-made and whether they constitute unwanted changes, such as environmental impacts or pollution.

Goals

One common problem of monitoring programs is that goals are often not clear and, therefore, sampling is often inappropriate to detect the changes that were the point of the exercise (Green, 1979). Unless it is clear what is to be monitored and why, preferably in relation to predictions (hypotheses) about the sorts of changes that might occur, sampling cannot be designed and implemented in cost-effective and statistically sound ways to achieve the supposed goals.

Thus, if the potential disturbance to some area of an estuary is the impact due to hydrocarbon leaks from marinas, the types and amounts of contaminating hydrocarbons can be specified, as can their locations and probable quantities. Designing sampling to detect such releases and any ecological consequences is then relatively straightforward.

If, in contrast, sampling is supposed to detect changes in some poorly defined concept (e.g., "ecosystem health"), no relevant variables are defined and no amount of anticipated deterioration in ecological variables can be identified. It is then impossible to determine what, where, and when to sample, let alone how to do so effectively. Nevertheless, such "monitoring" is commonly done.

Types of changes

Different types of study are required for different types of disturbances. For example, if the problems are due to chemical contamination, it is quite straightforward to design sampling to detect the background levels of chemicals. Provided the amounts of change to be detected are defined in advance, it is not difficult in principle to define sampling that is appropriate to identify such changes.

It is, however, less clear that measuring chemicals is a good way to identify impacts. Contamination is the presence of unnatural chemicals in the environment. Pollution is some biological or ecological response to the contamination. It is common for the concentrations of chemicals to increase without any particular response by organisms. It is equally possible for there to be no change in concentrations of chemicals, but for there to be a large-scale impact.

Consider the concentrations of nitrogen in estuarine water. Increased input of nitrogen can lead to rapid and excessive growth of algae in the plankton or sediments. Algae take up the extra nitrogen as they grow, resulting in no change in the concentration of nitrogen in the water column. Thus, monitoring for impacts on algae by measuring nitrogen is useless; the measure will not change regardless of the amount of impact of increased nitrogen. In this case, it would, instead, be necessary to monitor the algae themselves. Using indicators of required measures is often not sensibly planned.

Impacts occur in many forms. The most common are due to press as opposed to pulse disturbances (Bender et al., 1984). Press disturbances are those which last a long time. For example, building a marina will cause long-term changes to water flow, shading, etc. A pulse disturbance is a relatively short-term problem that then ends. For example, short-term dredging can cause increased sedimentation in surrounding areas, but this will stop when the dredging ceases.

An impact in response to a press disturbance can be a press impact, where the monitored variables change to a new average value and then show any natural temporal fluctuation around that value. Alternatively, a press disturbance can cause a pulse impact, where the monitored variables fluctuate in response to the disturbance. Pulse disturbances can also cause pulse or press impacts. A press impact would occur when the system being affected continues to be affected even though the original disturbance has ceased, for example, because a short-term disturbance causes long-term changes to a habitat. Identifying and measuring press as opposed to pulse responses to disturbances require different types of sampling (Underwood, 1994).

Spatial and temporal issues in monitoring

Spatial replication

Spatial replication is necessary in any sampling because of large variability in all ecological variables. Suppose amphipods are sampled in one area where dredging has disturbed the sediments and in one similar area where sediments have not been disturbed. Any difference found may be due to the disturbance or to *any* ecological processes affecting the numbers of amphipods differently in the two areas (Hurlbert, 1984). Any conclusion about an impact is confounded because the difference may also be due to natural variation.

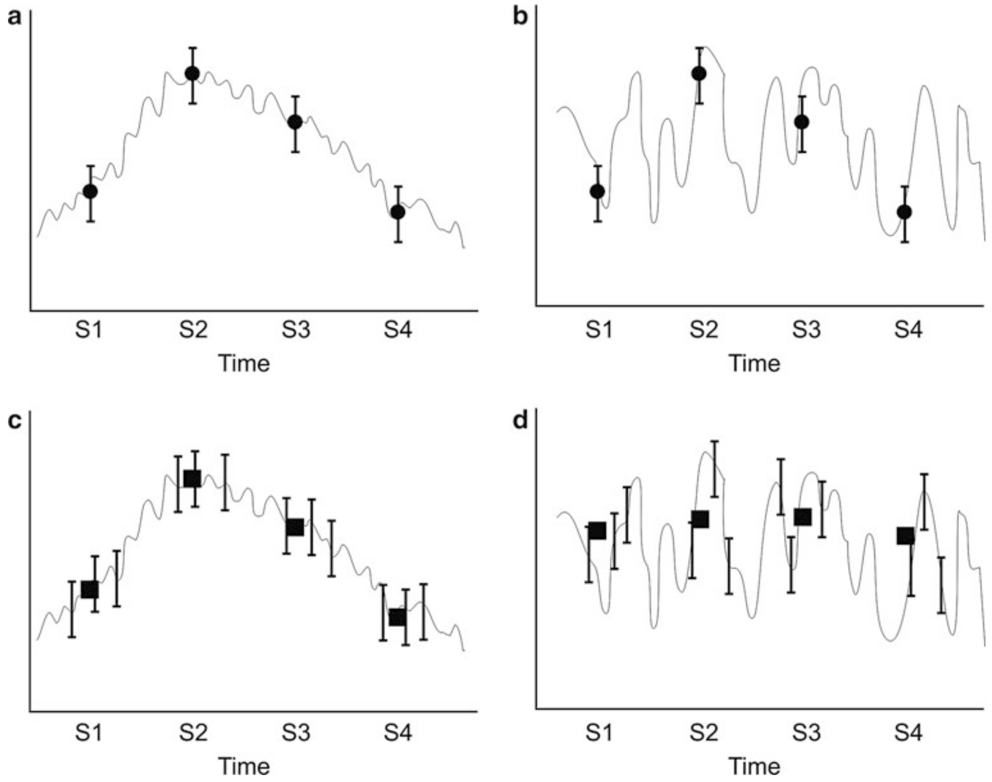
It is necessary to sample several dredged and several undisturbed areas so that natural variability that is not due to dredging can be measured under disturbed and control conditions. An overall difference between the dredged and undisturbed areas that is larger than natural variation among the areas of each type would then be an unambiguous evidence for an impact.

Spatial scales of sampling

Sampling at only one spatial scale can, however, cause problems. Suppose that several places in an estuary have been dredged, but the dredged sediments are contaminated by metals which may affect benthic infauna in areas around the dredged sites. Sediments are thought to carry metals up to 100 m away. Sampling is done in several dredged and several controls. In each area, the infauna are counted in replicate cores of sediment. Natural variation from replicate to replicate and area to area can then be estimated, and analysis could reveal any systematic difference between dredged and control areas.

Suppose, however, that contaminated sediments actually get dispersed over the entire area of the estuary, not just to sites within 100 m of dredged sites. Any impacts will now cause changes in benthic infauna in all of the sites sampled. There can be no apparent impact, because the controls are affected in the same way as are sites close to dredged areas. The interpretation would now be that some estuary-wide change had occurred to the infauna, which was not due to the dredging.

Other estuaries where dredging was not being done should also be sampled as controls. Uncertainty about the scale of potential impacts requires sampling at several scales. Analysis of data in such situations has been described in detail by Green (1979) and Underwood (1994).



Ecological Monitoring, Figure 1 With only a single sample (*black circles*) at a series of time intervals (e.g., each season), an apparent seasonal pattern can be identified in the variable being measured whether (**a**) there is, indeed, a long-term seasonal trend or (**b**) there is considerable short-term variability but no long-term trend. Short-term temporal sampling is needed within each season (sampling shown as vertical bars at each of the three times in each season). This provides the correct form of within-season replication to measure seasonal changes, and the means of the short-term results (*black squares*) distinguish (**c**) long-term trends from (**d**) background "noise".

Appropriate temporal replication

Monitoring to detect temporal changes requires replication to measure temporal variability (Stewart-Oaten et al., 1986). Seasonal patterns of abundance are often measured by sampling a number of replicates once in each season, using replicates taken at the same time in the sites sampled. The variation among replicates in each sample is *spatial* variation, because the replicates are all taken at one time, even though spatially scattered. Seasonal patterns are not being contrasted against temporal variation within each season.

To test for seasonal variation, seasonal differences must be compared to temporal variation *within* each time period of interest (Figure 1). It is essential to collect samples several times within each season.

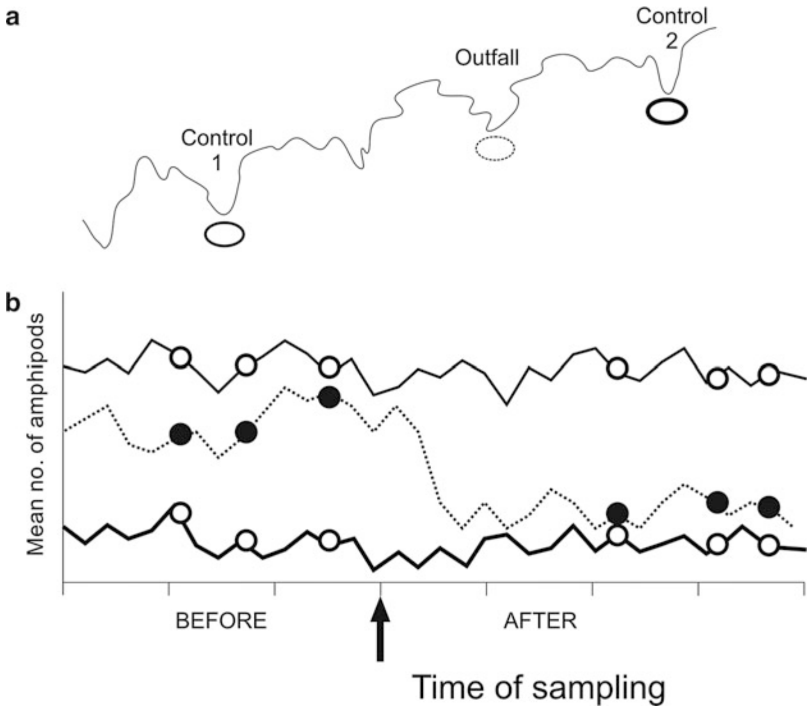
Impacts are statistical interactions

Green (1979) and Underwood (1994) described in detail how to detect different types of impacts. There should be data from before (i.e., a baseline) and after a disturbance that might cause impacts, so that an impact can be identified in sampling after the disturbance purported to have caused it. There must be proper temporal replication before and after the disturbance to provide reliable estimates of average conditions (Stewart-Oaten et al., 1986) and to estimate temporal variance, which might itself be altered by an environmental disturbance (Underwood, 1994).

There should be replicated, undisturbed controls to demonstrate that an impact is in the disturbed area and not a general phenomenon which is not due to that disturbance (Green, 1979). "Undisturbed" in this context means subject to any other influence or process except the particular disturbance under investigation.

An environmental impact will be detected as an ecological interaction, a change from before to after a disturbance which is not the same in the disturbed area as in the control areas where that disturbance does not occur. To analyze impacts, it is necessary to design sampling which will provide data that can be analyzed to detect and interpret statistical interactions.

One type of ideal design is illustrated in Figure 2 for potential impacts on intertidal algae on rocky shores due to the construction of a sewage outfall on a shore in an estuary. Two control areas with similar features of habitat (rocky headlands with similar currents, depth of water) were also sampled. Any change in algae that is not due to the discharge of sewage would affect the control areas and the outfall location.



Ecological Monitoring, Figure 2 (**a**) Sampling to detect impacts from the construction of coastal sewage outfalls. An outfall (····) and two similar control sites (——, ▬) are sampled three times (●, ○). Before and again after the outfalls begin to discharge. (**b**) Data that would indicate a large-scale, consistent "press" impact.

Data are collected three times (essentially chosen at random) over 2 years before the outfalls begin to discharge sewage and then three times over a period of 2 years, starting 2 years after the outfalls are commissioned (Figure 2). At each time of sampling, the amount of algae is recorded in a number of replicates. A large and consistent press impact (Bender et al., 1984), a shorter-term, more fluctuating pulse impact, or impacts causing changes in temporal variance can be analyzed (Underwood, 1994; Underwood and Chapman, 2013).

Precautionary principles and errors in interpretations of monitoring

Precaution has become a guiding goal for decision-making about environmental change (see Dovers and Handmer, 1995). There is always uncertainty about any information used in managerial decisions; therefore, mistakes can occur in the interpretation of analyses of data from monitoring. An impact could apparently be found when, in fact, there is none (in statistical tests of hypotheses, this is called a type I error). The opposite mistake occurs when no impact is found, even though one has occurred (a type II error). Where monitoring suggests that an impact has occurred, more work will usually be done to determine its extent and to test further the hypotheses about its consequences. The mistake will probably be detected. In contrast, if a real impact has not been detected, no particular action will be taken and environmental degradation will continue (Underwood and Chapman, 2013). The "cost" of type II errors is much greater than that of type I errors. It is therefore very important to consider, before any monitoring scheme is designed, how large, intense, or frequent any environmental change is likely to be. Then it is possible for (and precautionary principles require) the sampling be designed to maximize the chance of detecting real changes and to minimize the chances of missing real impacts (i.e., to avoid type II errors). This requires considerable skill and good statistical advice (Gray, 1996).

Conclusions

Monitoring is a necessary but complicated component of the environmental management of estuaries. It is used to detect any changes in physical, chemical, or ecological variables through time. The way the sampling should be designed is, however, totally dependent on what sort of changes are supposed to be detected and how intense, widespread, and frequent they are. Because of intrinsic variability in the things to be measured, sampling must be carefully designed to include appropriate and adequate spatial and temporal replication at the relevant spatial and temporal scales for the processes being examined. If this can be achieved, reliable interpretations can be made, leading to properly informed decision-making. Although the complexity of spatial and temporal scales is great, the methods available to analyze and interpret data obtained by monitoring are largely understood and widely available (e.g., Green, 1979; Spellerberg, 1991; Underwood, 1994; Clarke and Gorley, 2006; Schmitt and Osenberg, 1996). With care in the formulation of objectives and skillful advanced planning of the procedures, monitoring provides good information to enable logical and precautionary responses to environmental change.

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Cross-references

Anthropogenic Impacts
Climate Change
Coastal Indicators
Water Quality

ECOLOGICAL NICHE

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Synonyms

Hutchinsonian niche

Definition

The ecological niche comprises the total physical space within which the individuals of a species survive, grow, and reproduce. The dimensions or limits defined for this space constitute an abstract concept, not only a volume in space.

The niche concept is one of the most important concepts in ecology, although it is one of the most confusing (Root, 1967). The first general idea of the niche concept was proposed by Grinell (1917), who established it as a measure of the geographic distribution of an organism, without explicitly using the term niche. With time, new proposals have arisen, but in essence, in every case the niche is considered not just as physical space, it also includes the environmental factors that can affect organisms, the means that they use to obtain food, and their relations with other organisms (e.g., competition, predation) (Elton, 1966; Whittaker and Levin, 1975; Krebs, 1985).

The term niche describes not only where an organism lives but also how it lives (Townsend et al., 2008). The most accepted definition of niche was the one proposed by Hutchinson (1957), who stated that the ecological niche includes all physical and biological variables that affect the good functioning of an organism. This he called the multidimensional niche or hypervolume; this last term refers to upper and lower intervals, or thresholds of survival of the species within limiting variables. This puts in context the ways in which tolerance and requirements interact to define the conditions and needs that an individual or species have in order to live. It should be noted that different organisms have different tolerances to limiting conditions (e.g., temperature, pH, relative humidity, wind velocity, water flow, etc.), as well as different needs for several resources (e.g., water, nutrients, food, etc.), so that the niche concept is without doubt multidimensional.

It has been proposed that the fundamental niche of a species can change slowly under natural selection. Models have been developed that, along with population dynamics and genetics in heterogeneous environments, have resulted in predicting that the rate of adaptation of the fundamental niche would often be slower than the process of extinction.

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ECOLOGICAL STOICHIOMETRY

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Synonyms

Biological stoichiometry

Definition

The study of the interaction of chemical resources (elements) in organisms and the environment; the study of the balance of energy and materials in an ecosystem.

Introduction

Ecological stoichiometry is a rapidly expanding research area focusing on how the balance of chemical elements are related to community structure through differences in the apportionment of these elements in organisms and the rates by which these elements are recycled and mineralized in food webs (Sterner and Elser, 2002).

A new concept with historical roots

Ecological stoichiometry is a relatively recent conceptual framework for understanding the interactions of organisms in relation to energy and elemental flow. It builds on classical concepts of Liebig's Law of the Minimum relating to nutrient limitation (Leibig, 1855), Lotka's (1925) understanding of the dynamics of predators and prey, Lindeman's understanding of trophic dynamics (Lindeman, 1942), and Redfield's (1934) concept of balanced proportions of elements in the ocean. Ecological stoichiometry brings these concepts together by recognizing that different organisms both within and between trophic groups have fundamentally different elemental requirements, that food web structure is a function of not only food quantity but food quality, and that these interactions result in a complex suite of feedbacks that shape community composition. These relationships are linked via elemental composition of the interacting organisms. The stoichiometric framework recognizes that changes in the proportions of dissolved nutrients in the environment have profound effects on food webs even when the availability of these elements are not in limiting proportions, with the potential of transforming ecosystems to new stable states. In short, the concept of ecological stoichiometry suggests that, while the total nutrient load of a system may set the amount of biomass that can be supported, the composition, both in form of nutrients and the proportion of different nutrient elements, affects the composition of the community, from autotrophs to heterotrophs.

A focus on the major nutrient elements

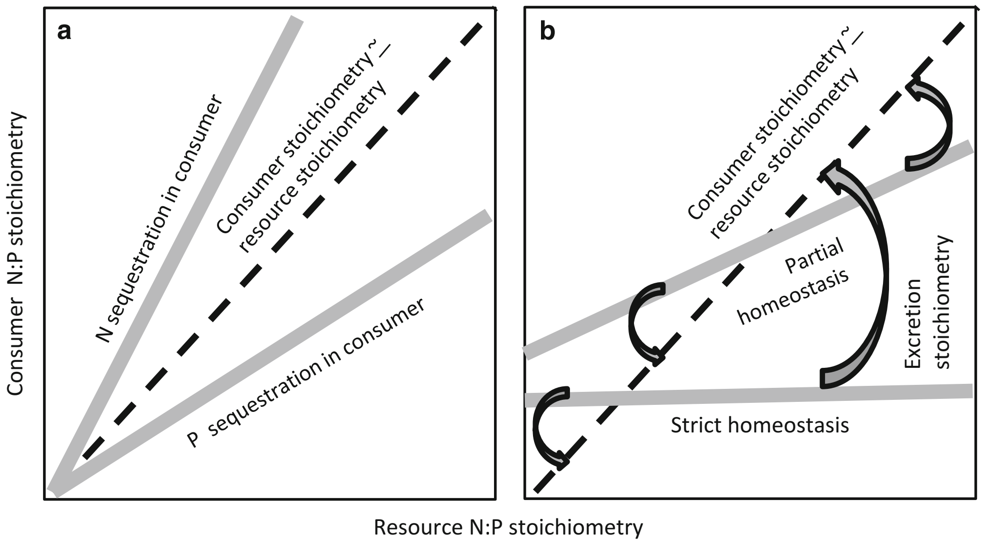
Of the naturally occurring elements, ecological stoichiometry is largely concerned with the major nutrients, carbon, (C) nitrogen (N), and phosphorus (P), although the fundamental principles can be applied to other elements such as silica (Si), iron (Fe), calcium (Ca), and other minor elements required for life as well. These elements comprise the major biochemical and organic molecules of which organisms are composed including nucleic acids, lipids, amino acids, proteins, pigments, carbohydrates, and skeletal components of larger organisms. Organisms diverge in their chemical needs for these elements. How they regulate their chemical composition affects their ability to survive and grow, and it also affects the organisms around them by the alteration of food quality for the next trophic level and by the recycling of the elements via regeneration and excretion.

Stoichiometry and primary producers

Primary producers, including algae, are considered to be comparatively flexible (within limits) in their stoichiometry; they often follow the "you are what you eat" model (Sterner and Elser, 2002; Figure 1a). It has been well established that many algae have the ability to take up nutrients in the proportion they are available, including an ability to take up nutrients in excess of their growth demands for certain periods of time, or as functions of varying temperatures, growth rates, and light conditions (e.g., Rhee, 1978; Glibert and Goldman, 1981; Finkel et al., 2010). This leads to a fairly wide range of variation in C:N:P ratios in algae (Klausmeier et al., 2004). Thus, although the Redfield ratio is often used to infer elemental composition in phytoplankton, the actual elemental composition of microalgae in culture and phytoplankton in nature is highly variable (Geider and LaRoche, 2002; Finkel et al., 2010).

Stoichiometry and consumers

In contrast to microbial primary producers, many heterotrophs exhibit much more "rigid" or "homeostasis" in the stoichiometry of their biomass (Figure 1b). They are comparatively more constrained in their C:N:P ratio. (Note that many microheterotrophs such as heterotrophic flagellates are excluded from this generalization.) Heterotrophs typically maintain a stricter stoichiometry because of the fixed stoichiometry in body tissue, especially muscle, bone, and other organs (Sterner and Elser, 2002), although they too may also be subject to some degree of variability, especially with regard to C versus nutrient elements (e.g., Malzahn et al., 2010). Grazers are able to stabilize their biomass stoichiometry more than phototrophs because they have excretion and release



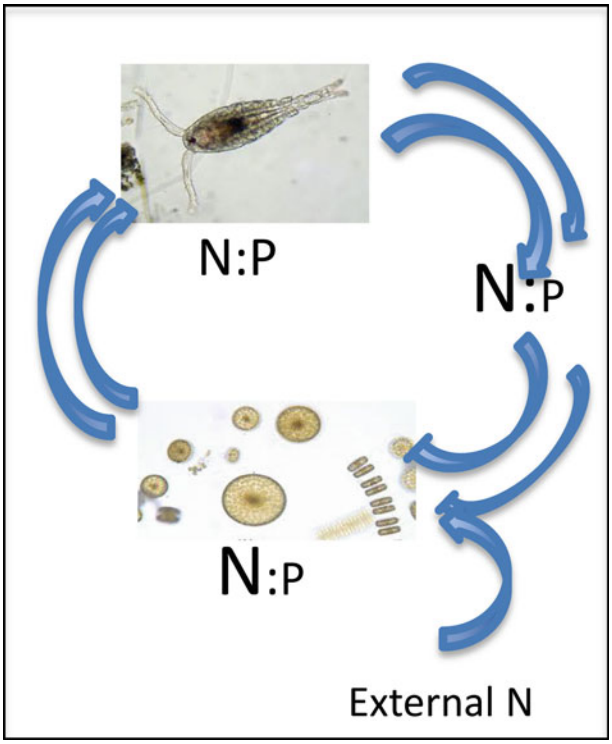
Ecological Stoichiometry, Figure 1 Schematic relationships between resource N:P (either dissolved nutrients or prey) and consumer N:P. (**a**) Hypothetical situations in which the consumer is either N or P enriched relative to its resource in a constant proportion. The *dashed line* in both panels represents the hypothetical situation in which the consumer N:P matches that of its resource. (**b**) Hypothetical situations where the consumer either partially or strictly regulates its biomass N:P regardless of the N:P of its resource. The *arrows* depict the extent to which the excreted or released nutrients differ in N:P from that of the consumer biomass N:P. Excretion N:P is expected to be negatively related to substrate N:P when the consumer N:P is constrained (Reproduced from Glibert et al. (2011), *Reviews In Fisheries Science* with permission of the publisher).

pathways to eliminate the nutrients that are not needed (Vanni, 2002). By excreting, egesting, or respiring what they do not need, an effective feedback develops with respect to the element stoichiometry of their resource or prey (Figure 1b).

Changes in nutrient proportions have ramifications at all levels of the food web. Consider, for example, the case of a consumer, a zooplankter, with a relatively low N:P biomass (Figure 2); in other words, such an organism would have a high P requirement. If they feed on food that has a higher N:P than their biomass demand, the zooplankter will disproportionately retain P and excrete N. This produces a nutrient pool that, after many turns of the cycle, becomes increasingly N:P enriched. If the primary producers are less constrained in their biomass stoichiometry, they will continue to reflect the composition of the external nutrient pool (not infinitely, but in a comparative sense). Thus, ecological stoichiometry principles would suggest that homeostasis from nutrient recycling will drive the nutrient balance of the system to be self-sustaining (Figure 2). Such principles further suggest that biodiversity should be a consequence of stoichiometry and that populations should self-stabilize as a result of stoichiometric constraints. Just as different elemental ratios may affect the composition of the primary producers, different nutrient requirements of higher trophic levels will have an impact on their ability to thrive as community composition changes at the base. As summarized by Sterner and Elser (2002, 263), “Stoichiometry can either constrain trophic cascades by diminishing the chances of success of key species, or be a critical aspect of spectacular trophic cascades with large shifts in primary producer species and major shifts in ecosystem nutrient cycling”.

Variation in stoichiometric regulation

Stoichiometric regulation of trophic interactions can be modulated in a number of ways. Nutrient stoichiometry and food quality can have differential effects on consumers depending on their life stage, e.g., larval or adult. Larvae would be expected to have higher P demands, for example, than adults, due to their higher growth rates (Boersma et al., 2008). Stoichiometric regulation can also be altered when “good food goes bad” (sensu Mitra and Flynn, 2009). Many attributes of food can be altered chemically and physiologically, leading to trophic interactions that would not be anticipated strictly on the basis of elemental stoichiometry. As an example, production of allelopathic compounds or toxins can alter trophic transfer. Yet to some extent, production of toxins or allopathic compounds in algae may also be under stoichiometric regulation. For example, N-rich toxins may be disproportionately produced when algal cells are P-limited (e.g., Granéli and Flynn, 2006). The dominance of toxic algae can result in a failure of normal predator–prey interactions, which in turn enhances the transfer of nutrients that sustain such species at the expense of competing algal species (Glibert, 1998; Sunda et al., 2006; Glibert et al., 2010).



Ecological Stoichiometry, Figure 2 Schematic of an algal assemblage that is relatively rich in N compared to P (driven by elevated external N loads), grazed upon by a copepod with a more balanced biomass stoichiometry. In such a case, the excretion products would be expected to be elevated in N:P. This, in turn, sustains the algae in an N-rich state.

Application to estuaries

Estuaries are highly dynamic systems with respect to nutrients. Nutrients are changing in estuaries due to nutrient pollution, on the one hand, but concerted efforts for nutrient reduction, on the other, at least in some systems. Although eutrophication is occurring globally, nutrient export from coastal watersheds is not evenly distributed, leading to highly varying N and P loads to coastal waters (Seitzinger et al., 2005; Glibert et al., 2006; Howarth, 2008). The consequence of these alterations in global N and P is that many receiving waters are now not only enriched with nutrients, but also these nutrients are being delivered in a different stoichiometric proportion than in decades past. Ecological stoichiometry applications in estuaries are even more complex because biogeochemical pathways, together with homeostatic control, serve to provide the mechanism(s) whereby nutrient dynamics support trophodynamic structure (Glibert, 2012). Positive reinforcing feedbacks of biogeochemistry and homeostasis shift ecosystems to new stable states; such shifts can be gradual or abrupt, and communities may not return to their original state once the disturbance (in this case, altered nutrient loads) is removed. In eutrophic systems, increased algal productivity may lead to depressed water column oxygen which, in turn, may result in increased recycling of N and P by changes in redox potential or pH (Kemp et al., 2005; Glibert et al., 2011; Glibert, 2012; Gao et al., 2012). These fluxes will then positively reinforce an ecosystems degradation trajectory, as suggested to be the case for the Chesapeake Bay (Kemp et al., 2005). In contrast, in systems with a smaller nutrient load, less algae in the water column, higher light, and higher redox potential may help to reinforce higher rates of nitrification and denitrification, leading to nutrient removal and potentially eutrophication reversal (Kemp et al., 2005).

Summary and conclusions

In sum, ecological stoichiometry theory describes how food quality affects food web dynamics by defining not only the pathway of flow of needed elements by consumers but also the pathways by which “excess” nutrients are recycled, further altering nutrient availability for lower trophic levels. Ecological stoichiometry has several important implications for the health and sustainability of aquatic systems. Stoichiometric imbalances may accelerate transformations of nutrients or may alter the processes by which nutrients are cycled in the ecosystem and thus nutrient availability or form for primary producers (Elser and Hamilton, 2007). When food *quality* is linked to food web outcome, feedback effects and nutrient biogeochemical processes may play large roles in species success. Moreover, ecological stoichiometry bears significantly on the debate of whether aquatic ecosystem restoration efforts should focus on P removal, N removal, or both (e.g., Carpenter, 2008; Conley et al., 2009; Doney, 2010 and references therein). Single nutrient removal strategies can drive ecosystems into states of stoichiometric imbalance. Imbalances in stoichiometry may destabilize the dynamics of consumers, shifting systems to new conditions. Single nutrient removal strategies may have unintended consequences for aquatic ecosystems.

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Cross-references

Algal Blooms
Eutrophication
Food Web/Trophic Dynamics
Trophic Dynamics

ECOLOGICAL SUCCESSION

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Synonyms

Biotic succession; Community succession; Ecosystem succession; Succession

Definition

Ecological succession refers to the sequential replacement of species, communities, or ecosystems over time. When applied to a given site, succession can encompass change in communities (often defined by their dominant species) and ecosystems (which drive or are driven by species change), structural and functional change (usually but not always driven by the biota), and abiotic environment change.

The ecosystem is dynamic. Ecological succession has internal and external causes. The interaction between various components within an ecosystem is the primary internal driving factor. External driving factors are natural (e.g., land changes and tsunamis) and man-made (e.g., deforestation and fishing) which add to the ecological system. The external factors only can work through the internal factors. The mechanisms include dispersal and establishment of species, competition, and facilitation. Stages can be classified into pioneer or initial, early, mediate, and late stage. The final state of dynamic balance is called climax.

Ecological succession can be divided into progressive succession (generally following a period of increasing biodiversity, biomass, and active nutrient cycling) and retrogressive succession (generally following a period of decreasing biodiversity, biomass, and nutrient cycling).

The succession which commences with a barren shoal in an estuary is called primary succession. The secondary succession occurs after the damage of an original ecosystem. Succession does not incorporate regular disruptions such as seasonal changes or very long-term processes.

Succession is rarely predictable at the level of species composition but is sometimes predictable at the level of functional groups of species. The principle of ecological succession can guide ecological restoration.

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ECOSYSTEM-BASED MANAGEMENT

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Synonyms

Ecosystem approach to management; Integrated natural resource management

Definition

Ecosystem-based management (EBM) is an integrated approach to management that considers the entire ecosystem, including humans. The goal of ecosystem-based management is to maintain an ecosystem in a healthy, productive, and resilient condition so that it can provide the services humans want and need. Ecosystem-based management differs from current approaches that usually focus on a single species, sector, activity, or concern; it considers the cumulative impacts of different sectors (COMPASS, 2005).

History

Until the last few decades, the predominant way to approach managing sections of the Earth was to focus narrowly on particular aspects of an ecosystem (e.g., managing a forest for production of lumber or for the protection of a particular animal species). The seeds of a new way of looking at ecosystem management can be found as early as the 1950s, in the writings of Aldo Leopold among others (Arkema et al., 2006). During the late 1960s, scientists and policy scholars in the United States began to connect the traditional and narrower approach to management with a noticeable downturn in the health of both individual species and other measures of ecosystem health. These critics hoped that the passage of laws such as the National Environmental Policy Act of 1970 would influence management to be more holistic and to take into account a greater number of factors than the single-species approach (Layzer, 2012). This new approach began to be characterized as the "ecosystem approach" (Slocombe, 1998), and the 1970s saw two high-profile efforts to begin to manage systems with this broader paradigm, in the Great Lakes and in the Chesapeake Bay (Slocombe, 1998; Layzer, 2012), though most authors tend to point to the Great Lakes example, which resulted in the Great Lakes Water Quality Agreement of 1978 – as the most comprehensive and seminal effort in the early history of EBM.

In the 1980s, two new high-profile issues erupted, both of which brought more public and scientific focus to the issue of an ecosystem approach to management. The first issue involved disputes about how to manage the grizzly bear habitat in the Greater Yellowstone region; the second involved the campaign to protect the northern spotted owl in the forests of the Pacific Northwest (Layzer, 2012). These two issues were notable for several reasons: (1) they were highly contentious and well-publicized issues that garnered a lot of attention in the national media; (2) both issues explicitly acknowledged that narrow management strategies were doomed to failure; and (3) both issues – especially in the case of the Pacific Northwest – also acknowledged that the interests of humans had to be considered as part of the management approach.

In the 1990s, EBM began to be invoked as a best practice strategy by many professionals and management agencies in the United States (e.g., US Forest Service, EPA, etc.) as well as across the globe. Although the term lost favor among land managers by the new millennium – due to pushback from those who equated an ecosystem focus with a left-leaning environmentalist movement – the ideas behind EBM remain strong among natural resource managers (Layzer, 2012).

The marine/estuarine focus on EBM began in the late 1990s and became solidified after both the Pew Oceans Commission (2003) and the US Commission on Ocean Policy (2004) explicitly called for EBM in addressing issues related to marine and estuarine waters. In 2006, commissioners from both efforts joined to create the US Joint Ocean Commission Initiative, releasing a report that focused on EBM as critical to achieving many of the objectives (Arkema et al., 2006). In contrast with land resource managers, marine resource managers use the term "EBM" explicitly while admitting that the term is ambiguous (Arkema et al., 2006); many also note that the ideas are extremely difficult to actually put into practice (e.g., Boesch, 2006; Ruckelshaus et al., 2008), and actual cases of EBM occurring are rare and, arguably, Utopian.

Key characteristics (what EBM is/what EBM is not)

EBM is an approach to resource management that seeks to correct the deficiencies of traditional approaches that were seen as overly narrow. To paraphrase Slocombe (1998), EBM is about finding institutional, administrative, and scientific ways of managing whole ecosystems instead of small, arbitrary management units. A discussion of key characteristics for EBM must explicitly differentiate EBM from "ecosystem management." EBM acknowledges that humans manage other humans and their impacts much more than they manage the nonhuman parts of the ecosystem. In contrast, ecosystem management was more liable to ignore the human dimension aspects of ecosystem issues (Slocombe, 1998; NRC, 1999; Boesch, 2006; Layzer, 2012), tending to be dominated by ecological science to the exclusion of social sciences.

While there are many conflicting definitions and criteria for EBM (Arkema et al., 2006), most scholars and practitioners agree on a few basic principles:

1. EBM approaches generally occur at broad spatial scales, such as the Kluane Region of the Yukon in Alaska or the Great Lakes, which encompasses several states and two countries.
2. Humans and human activities are explicitly accounted for as part of the ecosystem, and human stakeholders are involved in the process of defining problems and addressing issues.
3. EBM approaches seek to understand linkages between multiple components of an ecosystem, rather than one species.
4. EBM approaches are constructed to be adaptive and flexible, expecting surprises and unforeseen consequences.

Arkema et al. (2006) emphasize that even isolating commonalities in EBM definitions at this general level can be difficult. For example, Layzer (2012) includes stakeholder involvement in management processes, but does not emphasize defining systems according to ecological boundaries as opposed to political ones (Levin and Lubchenco, 2008). A synthesis by Boesch (2006) mentions the multiple-component criterion but does not emphasize the human dimension component. Moreover, after the general principles of EBM are covered, scholars disagree on a whole host of supporting requirements for EBM. Arkema et al. (2006) reviewed 130 academic articles on EBM and extracted 17 different criteria, including interdisciplinary approaches, consideration of the precautionary principle, and integration of economic factors into decisions.

Estuarine-marine EBM

In estuarine marine EBM, as noted earlier, much of the recent activity that explicitly references EBM happens in the marine/estuarine context. In 2005, the organization, COMPASS, released a consensus statement defining marine EBM and its five emphases:

- Emphasize the protection of ecosystem structure, functioning, and key processes.
- Is place-based in focusing on a specific ecosystem and the range of activities affecting it.
- Explicitly account for the interconnectedness within systems, recognizing the importance of interactions between many target species or key services and other nontarget species.
- Acknowledge interconnectedness among systems, such as between air, land, and sea.
- Integrate ecological, social, economic, and institutional perspectives, recognizing their strong interdependences.

Leslie and McLeod (2007) outline four principles of marine EBM: (1) it is necessary to address multiple spatial and temporal scales; (2) it includes a focus on the link between marine and human communities; (3) it connects marine systems and management components with those focused on land and air; and (4) it emphasizes meaningful engagement of stakeholders.

Ruckelshaus et al. (2008) offer six principles that overlap with the above but are more operational and less philosophical. These principles are:

- Define spatial boundaries of the system to be managed.
- Develop a clear statement of the EBM objectives.
- Include humans in characterizations of marine systems and indicators of change.
- Use a variety of strategies to hedge against uncertainty in the ecosystem response to EBM approaches.
- Use spatial organizing frameworks such as zoning for coordinating multiple management sectors and approaches in EBM.
- Links the governance structure with the scale of the ecosystem elements to be managed under an EBM approach.

Like other scholars, Ruckelshaus et al. (2008) note that actual applications of EBM are relatively rare, yet they do offer some examples of partial success. These examples can serve to better illustrate the potential benefits of EBM. One of these examples comes from the waters of the Southern Ocean surrounding Antarctica. Since 1982, management of marine resources in this area has been supervised via the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR). The krill fishery is one of the most significant activities needing management in this area. In a traditional approach, management of the krill fishery would have been concerned with the continued survival and sustainability of the krill populations alone. In an EBM context, however, other species and system components must be considered. Therefore, fisheries models were developed to account for negative impacts to various predators on krill as well as the krill themselves, resulting in recommended rates of removal that were 25 % lower than in a more traditional and narrow approach.

Another example discussed by Ruckelshaus et al. (2008) involves the Great Barrier Reef off the coast of Australia.

In this example, managers have instituted several notable strategies that characterize the marine EBM ideal. First, authorities established no-take zones throughout the area, endeavoring to represent a diversity of key habitat types. In addition, managers were aware that land use decisions and activities have a significant impact on the Great Barrier Reef system and therefore created mechanisms for the regulation of some of these activities.

The authors note, however, that even in these rare cases of partial EBM implementation, there is considerable room for improvement. For example, in the case of the Great Barrier Reef, many of the social science questions (such as the economic impacts of the EBM approach) have not been researched and integrated into the plan. In addition, scientists acknowledged that the impacts of land-based inputs of sediment and nutrients from development are still poorly understood.

Challenges to application

Readers will not be surprised that those who study EBM implementation universally mention how challenging it is to actually accomplish even some of the EBM principles, let alone all of them. At the heart of EBM is the injunction to think much more broadly – both ecologically and operationally – about the management of ecosystems. Broader efforts are, by definition, more complicated than narrower ones as they increase the sheer number of factors to consider as well as the number of factor types, requiring higher numbers of disciplines and perspectives. However, researchers also point out more specific and less obvious reasons why EBM implementation is so challenging.

Slocombe (1998) notes that many obstacles to EBM are also obstacles to other endeavors (e.g., integrated resource management). Slocombe's list of obstacles includes (1) too much fragmentation in the administration and research sectors, (2) too much competition for resources within/between various agencies, and (3) obscurely articulated objectives (e.g., "sustainability") in the planning process.

Boesch (2006) uses two case studies (Chesapeake Bay Estuary and Coastal Louisiana) to focus on the scientific requirements and challenges of EBM. He suggests several challenges that must be overcome in order for scientific contributions to better impact resource management of ecosystems. Two of these challenges are discussed here. First, scientific organizations must increase the emphasis on using science to guide and test solutions, rather than simply continuing to characterize the state of ecosystems. Boesch is careful to state that curiosity-driven research should be continued, but he makes it clear that the balance needs to be adjusted so that greater priority is given to solution-oriented science. Secondly, society needs to invest in more bridge organizations and bridge activities. The intent is not, Boesch stresses, to merge the science world with the decision-making world; rather, society needs more institutions and people who can act as effective bridges between the two worlds. For this to happen, Boesch asserts that the financial incentives will have to change so that integrative work becomes valued more in the same way that specialization work is currently valued, both in the science and nonscience sectors.

At this point, it is important to take a step back and reassess the role of science in managing natural resources. Boesch's work begins by noting that both the Pew Ocean Commission and the US Commission on Ocean Policy urged modifications in the way science is done in light of the EBM approach. However, it should be noted that both these reports consider science to be only one of several prerequisites for better resource management; in other words, better science alone – without other significant changes – will not lead to better resource management.

Among those professionals that work in the field of linking science to decisions – conducting the "bridge" work referred to earlier – there is a growing consensus that the role of science is often both overestimated and underestimated. It is overestimated in the sense that participants often expect rational and scientific processes to dominate resource management questions when, in fact, issues related to culture and values are usually more predictive on how people make decisions. Gregory et al. (2006) note that making good decisions requires good science, but it also requires an acknowledgement of the power of culture and values. Cash et al. (2003) note that issues related to trust and relationships are as important as perceptions related to the technical quality of the science. Finally, the role of science is underestimated in that many in society think overly narrowly in their conception of "science," forgetting that rigorous scientific methods exist and can be applied to incorporating values, addressing problems related to different perceptions of risk (Gregory et al., 2006), and studying how to conduct science in a way that nurtures existing relationships (Matso and Becker, 2013).

Layzer (2012) also focuses on the issue of how to marry good science, stakeholder engagement, and good decision making. She notes that there is debate and ambiguity about the role of collaborative stakeholder processes in the development and implementation of EBM approaches. Specifically, some studies clearly prioritize protection of ecosystem integrity, using stakeholder engagement as a means to gather local knowledge and buy-in in order to achieve the goal of conservation. Others, however, emphasize that EBM is about actually achieving two goals simultaneously: ecosystem integrity and social/economic goals of the human communities. These disparate interpretations of EBM are quite different and would result in very different implementation strategies.

Layzer also conducted a systematic assessment of EBM in practice, using seven case studies as her main source of data. Although all seven case studies showed the promise of environmental benefit, given enough time, Layzer found that some of the cases – especially those that relied on consensus building in their collaborative process – produced results that were less likely to provide significant benefit to the environment. Layzer notes that, in these cases, consensus building incentivized dodging tough trade-offs by trying to provide something for everyone and results were less protective of the environment.

It should be noted that "consensus building" is only one way of approaching collaborative decision making. Other approaches, such as Structured Decision Making (Gregory et al., 2012) and Collaborative Learning (Daniels and Walker, 2001), explicitly note that consensus is not always the correct method in collaborative processes. Sometimes it can lead to less creative solutions.

Layzer (2012) also noted that successful (i.e., most likely to show environmental benefit) examples of EBM – looking beyond her seven cases – tend to involve contexts that corresponded to Ostrom's criteria (1990). These criteria include:

- Participants believe they will be harmed if they do not adopt rules for resource management.
- Participants are affected in similar ways by the proposed rules.
- Participants value the continued use of the common property resource.
- Participants have relatively low hurdles in terms of obtaining new information, making changes, and enforcing rules.
- Participants share generalized norms of reciprocity and trust.
- The group is relatively small and stable.

However, Layzer (2012) points out that the typical EBM project occurs at a large enough scale that many of these criteria are difficult to fulfill, which results in a very challenging situation for those trying EBM approaches. Another critical challenge to EBM is evaluating success. For example, Layzer (2012) discounts process-based evaluations that rely on stakeholder testimony about gains in trust and understanding. Rather, she focuses on data that show environmental benefit. However, this approach is controversial in and of itself. Many evaluation scholars (e.g., Conley and Moote, 2003; Mandarano, 2008) note that it is very difficult and possibly unwise to rely on environmental parameters in evaluating collaborative management efforts, because it is usually very difficult to isolate and show conclusively that an intervention did or did not have an impact with so many uncontrollable factors at play. In light of all these factors, most evaluators admit that using proxies such as stakeholder testimony regarding process and social outcomes (e.g., new rules and regulations) must be part of an evaluative best practice.

Summary

EBM remains a very relevant yet controversial topic in the arena of natural resource management. Society has made significant progress in understanding what EBM is and what it requires; however, truly successful implementation of EBM that succeeds in meeting all the principles of the concept is either extremely rare or nonexistent. Nevertheless, it remains – to most scholars and decision makers – a worthwhile alternative when compared with the traditional approach of single-species and single-purpose planning. The challenges of the future will be in understanding what sorts of institutional arrangements will be necessary to truly implement EBM and to evaluate it so that modifications can be made in the face of unintended consequences.

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Cross-references

Adaptive Governance
Adaptive Management
Precautionary Principle

ECOTOURISM

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Synonyms

Ecological tourism; Ecotourism

Definition

The business of organizing holidays to places that people do not usually visit in a way that helps local people, does not damage the environment, and can contribute to its conservation (Cambridge Dictionary, modified).

Fundamentals

Tourism is a sector of the world economy that provides a major contribution to growth and development, occurring in all continents (Harris et al., 2002; Hall and Page, 2006), even in the most remote in Antarctica, and shortly it will have its début in space. In this sector, as in many others of modern economy, new trends appear and bloom frequently. Some of these trends link to emerging and/or dominating social paradigms (Harris et al., 2002; Hall and Page, 2006). Such is the case of ecotourism that started to develop by the middle of the twentieth century (post-World War II), when tourism began to specialize in a number of different branches (coastal tourism, business tourism, sports tourism, third-age tourism, etc.) and, on the other hand, society became aware of environmental issues at large. In this way, people wanted to see and enjoy the disappearing forests, animals, ocean depths, and traditional people from distant lands and islands. In the collective, shared unconscious, it is possible that by witnessing natural wonders, people will be able to save the biota from extinction. At present, it is evident that we have permanently changed nearly half of the natural landscapes.

In effect, ecotourism activities can have many different forms and can be promoted in obvious spaces such as protected areas (EAA, 2013; IES, 2013) but also within urban environments. It is a field of the imagination and of entrepreneurship. However, to target and maintain sustainable practices is difficult to achieve. This means that it is a kind of tourism practiced at smaller scales, by small groups at a time, in a truly sustainable way, and involving local communities.

Increased attention has been given to this type of tourism since the 1990s, with many private and nonprofit groups exploiting ecotourism as their main income source for maintaining traditional ways of life. Governments are usually responsible for licensing, rating, and regulating this activity.

Many ecotourism activities involve water and coastal environments. Beaches and estuaries are important targets for ecotourism, due to the natural (and sometimes incalculable) values they possess and the communities that live and depend on the resources and services generated by those ecosystems. Several activities can be related to coastal/estuarine ecotourism:

- Animal watching
- Plant observation
- Fossil observation
- Flora and fauna conservation programs
- Trekking, climbing, and diving
- Photographic safari
- Archeology

There is a code of conduct for the practitioners of this type of tourism that includes:

- Minimization of impacts on the environment where it takes place
- Building environmental and cultural awareness and mutual respect
- Providing positive experiences for visitors and local communities
- Producing outcomes for environmental conservation
- Providing financial benefits and empowerment for local people, especially women
- Raising environmental awareness to host communities

One of the main goals of ecotourism is to deliver information and educate tourists about options of nature conservation in order to respect cultural and geographical differences. Therefore, it helps to develop/improve an ecological conscience, with a more sensible behavior toward nature, encouraging more involvement among tourists so that environmental values can be perceived and protected. Also important is that ecotourism has to be sustainable, and the local communities must be allowed to receive the majority of the social, educational, and economic benefits generated by this kind of tourism.

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Cross-references

Beach Management
Estuary Conservation

EMERGENT SHORELINE

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Synonyms

Emerged coasts; Emergence; Raised beaches; Uplifted shorelines

Definition

Emergent shorelines and coasts are those in the process of being lifted out of the sea.

Emergence and *emergent coasts* occur when relative sea level falls exposing new land. This process leaves *raised beaches* or *uplifted shorelines* on the land lifted out of the sea or exposed by a fall in sea level. Scandinavia, Scotland, and Canada are all classical areas of long-term emergence due to glacial isostatic uplift after the last ice age (Mörner, 1979). The sequence of raised beaches may take the form of continuous "staircases" of emerged beaches such as in the Hudson Bay (Hillaire-Marcell and Fairbridge, 1978). The highest glacial isostatically raised shoreline is in Sweden at +294 m and dates at about 9300 C14 years BP. Seismotectonics may also generate sequences of uplifted shorelines such as the famous Turakirae shorelines in New Zealand (Wellman, 1967). Coasts bordering subduction zones usually experience crustal uplift and seismotectonics, giving rise to raised beaches and emergent shorelines. This is the case in many parts of Japan and along the Pacific coasts of North and South America. In areas like New Guinea and Barbados, the uplift has led to sequences of raised coral reefs representing different interglacial sea level high-stands.

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ENDEMIC SPECIES

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Synonyms

Indigenous species; Native species

Definition

Endemic species are those that are unique or restricted to a particular locality or geographical area and cannot be found elsewhere.

The geographical location where endemic species are found can vary in scale, including small habitat patches or a defined zone, an island, or nation. Estuaries represent the confluence of terrestrial, freshwater, and marine ecosystems creating large environmental variation and fluctuations, which shape local adaptations and differentiation of taxa resulting in endemic species (Greenberg, 2012). Factors that influence differentiation and endemism include (1) abiotic stressors that provide opportunities for specialized functions, (2) degree of isolation from habitats (current and in the past), and (3) local changes in the habitat over time. Estuarine endemic species run a higher risk of extinction because of their restricted area and pressure from human alterations that include coastal development and modifications.

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Cross-references

Exotic Species
Introduced Species

ENVIRONMENTAL GRADIENTS

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Definitions

Environmental gradients in estuaries refer to gradual changes in abiotic factors and, consequently, in biotic factors, through space and time.

Ecotone is a transition area between two different habitats, ecosystems, or biomes.

Ecocline is a gradation from one ecosystem to another when there is no sharp boundary between the two.

Introduction

Estuarine ecosystems are characterized by a constantly changing mixture of saltwater and freshwater (McLusky, 1989). The flux of saltwater and freshwater presents physiological challenges to estuarine organisms. Gradients of salinity, for example, restrict many animal species in the adjacent sea or rivers from entering estuaries. In highly stratified estuaries, denser marine water entering an estuary along the bottom gradually mixes vertically with the outgoing surface flow of freshwater. Environmental gradients in estuaries depend on seasonal and daily fluctuations of abiotic factors.

Environmental gradients

Abiotic factors

Fairbridge (1980) defined an estuary as an inlet of the sea reaching into a river valley, as far as the upper limit of tidal rise, usually being divisible into three sectors: (1) a marine or lower estuary, in free connections with the open sea, (2) a middle estuary subject to strong saltwater and freshwater mixing, and (3) an upper or fluvial estuary, characterized by freshwater but subject to strong tidal action. The limits between these sectors are variable and subject to constant seasonal changes in the river discharge.

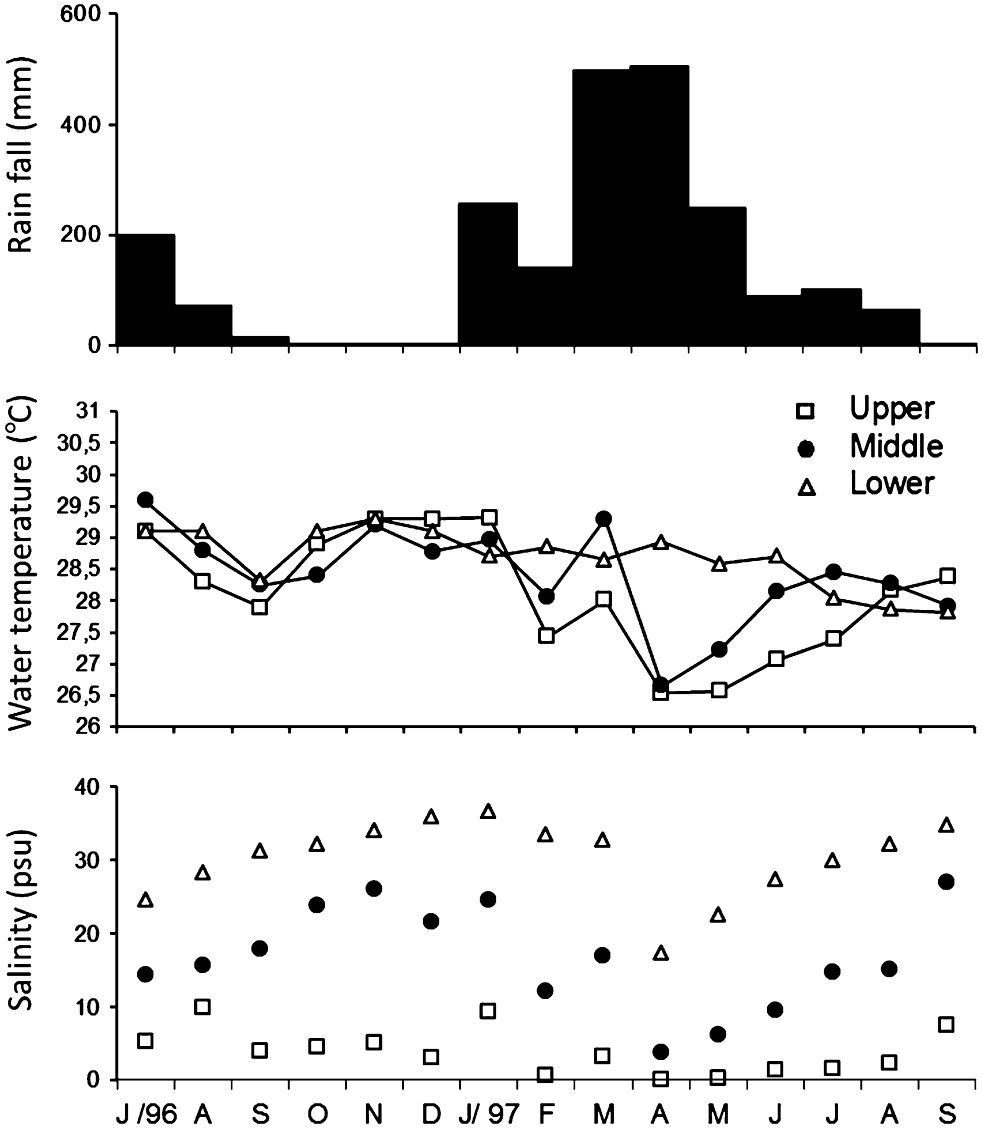
Estuaries may be classified as positive, neutral, or negative depending on their salinity regime and the extent of evaporation (McLusky and Elliott, 2004). In positive estuaries, the evaporation from the surface of the estuary is less than the volume of freshwater entering the estuary from rivers and land drainage. In such an estuary, the outgoing freshwater floats on top of saline water entering from the sea. Therefore, the water gradually mixes vertically from the bottom to the top in the estuarine basin. This type of estuary is thus characterized by incoming saltwater along the bottom, with gradual vertical mixing leading to an outgoing stream of fresher surface water (salt wedge). In negative estuaries, evaporation from the surface exceeds the freshwater runoff entering the estuary, and evaporation causes the surface salinity to increase. The saltier surface water is then denser than the water underneath, causing it to sink. The circulation pattern is thus opposite to that of a positive estuary because, in a negative estuary, the seawater and freshwater both enter the estuary on the surface, but after evaporation and sinking, they leave the estuary as an outgoing bottom current. Finally, in neutral estuaries, freshwater input equals evaporation and, in this case, a static salinity regime occurs. Depending on the tidal amplitude and volume of freshwater flow, four main types of positive estuaries are recognized: highly stratified, fjords, partially mixed, and homogeneous (Dyer, 1973, 1974).

Along an estuarine gradient, there are definite changes in salinity ranging from seawater to freshwater and associated changes in sedimentary conditions from coarse sediment (sand or gravel) outside the estuaries to fine sediments (mud) within the estuaries (Miranda et al., 2002). The horizontal distribution of average salinity in estuaries typically varies between 1 and 36, indicating that the saltwater was gradually diluted by the freshwater discharge. Moreover, the mixing and layering processes of salinity in the estuary depend on its geometry, freshwater discharge, tidal range, salinity, circulation of the adjacent oceanic region, and wind acting directly or indirectly on its surface. Other possible changes in estuarine gradients are alterations in turbidity of the water column or chemical composition, including changes in nutrients, dissolved gases, and trace elements (McLusky and Elliott, 2004).

Biotic factors

Estuarine organisms and habitats often experience steep gradients in salinity. Estuarine fauna must adapt to the dynamic environmental conditions. Many fishes, for example, have the physiological capacity to tolerate high fluctuations of salinity along an estuarine gradient. Estuarine fauna can be classified based on their salinity tolerances (McLusky and Elliott, 2004). For instance, oligohaline organisms are those that live in rivers/freshwater and do not tolerate salinities greater than 0.5. Most fauna with marine affinities that live in the central segment of estuaries (salinities of 5–18) are the truly estuarine forms. They can live in the sea, but the estuarine waters provide space and food resources as well as less competition with other marine species. Euryhaline marine species constitute the majority of organisms living in estuaries with their spatial distribution ranging from the sea up to the central segment of estuaries. Each species has its own range of salinity tolerance. Stenohaline is a term describing organisms that cannot tolerate a wide fluctuation in salinity. They are often fish and crabs that spend only part of their life cycles in estuaries, using these waters as pathways to and from their breeding areas in rivers or the open sea. The composition of estuarine fish assemblages is determined by a combination of biotic and abiotic factors, particularly competition for space and food, tolerance of diel and seasonal changes in salinity and turbidity, and temperature gradients (Barletta and Blaber, 2007).

Many studies on fish assemblages in estuaries and their relationships with habitat types and environmental gradients demonstrate that geology, geomorphology, and more immediate environmental gradient conditions, such as salinity and temperature, are associated with fish distribution, species richness, and fisheries catch (Mathieson et al., 2000; Thiel et al., 2003; Barletta et al., 2005; Barletta and Blaber, 2007; Barletta et al., 2008). For example, the relative proportions of freshwater and marine species using estuaries may be different depending on environmental gradients of salinity and other physical-chemical and geomorphological factors (Barletta et al., 2000, 2003, 2005, 2008; Blaber, 2000). Many functions of estuarine fauna, such as breeding, recruitment, nursery, and food supply, are subject to diel and seasonal fluctuations of environmental gradients. In neotropical estuaries (Figures 1, 2, and 3), the variation in the seasonal ecocline,



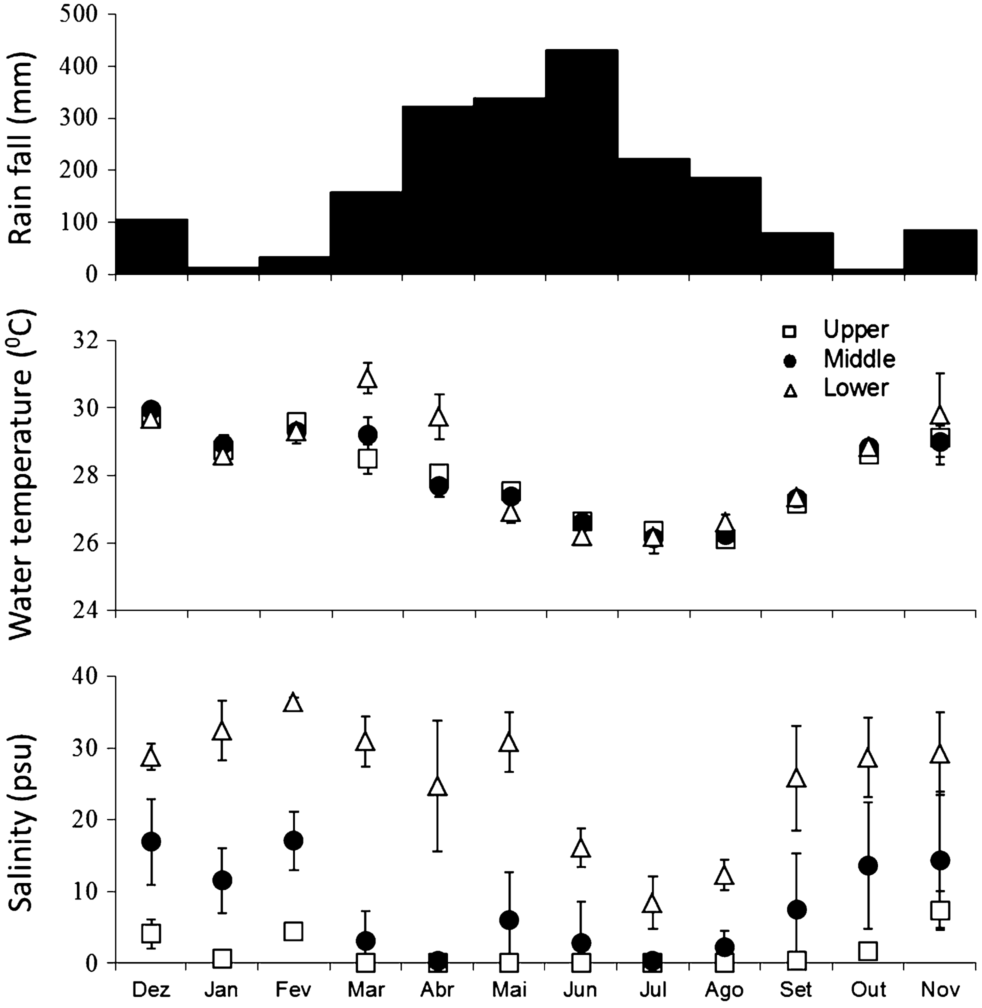
Environmental Gradients, Figure 1 Total rainfall (mm) and mean values of water temperature (°C) and salinity in Caeté Estuary (upper, middle, and lower estuary).

principally the salinity gradient, controls the distribution and movements of the estuarine fish assemblages along a spatial gradient (Barletta et al., 2005, 2008; Dantas et al., 2010).

Studies in neotropical estuaries of South America (Barletta et al., 2005, 2008; Dantas et al., 2012) suggest that for the catfish species *Cathorops spixii* (Agassiz) and *C. agassizii* (Eigenmann & Eigenmann), salinity gradients and dissolved oxygen levels influence the seasonal distribution not only of adults but also of all the different ontogenetic phases, along the estuarine ecocline (upper, middle, and lower estuary). The seasonal fluctuations in salinity (late dry and late rainy) and dissolved oxygen (early rainy and late dry) (Figures. 1, 2, and 3) define the middle estuary nursery area for *C. spixii* and *C. agassizii* (Barletta et al., 2005, 2008; Dantas et al., 2012). The importance of this habitat as nursery for *C. spixii* and *C. agassizii* juveniles is determined by the strength and position of the seasonal environmental gradient along the estuarine ecocline (Dantas et al., 2012). Estuarine ecosystems provide critical habitats for many commercial and recreational species (mainly feeding, mating, spawning, and nursery grounds), and they exhibit dynamic variability of environmental conditions.

Ecotone and ecocline

Ecotones are areas of relatively rapid environmental change, producing a narrow ecological zone between two different and relatively homogeneous community types. Ecotones are highly dynamic and usually unstable,



Environmental Gradients, Figure 2 Total rainfall (mm) and mean (±sd) values of water temperature (°C) and salinity in Goiana Estuary (upper, middle, and lower estuary).

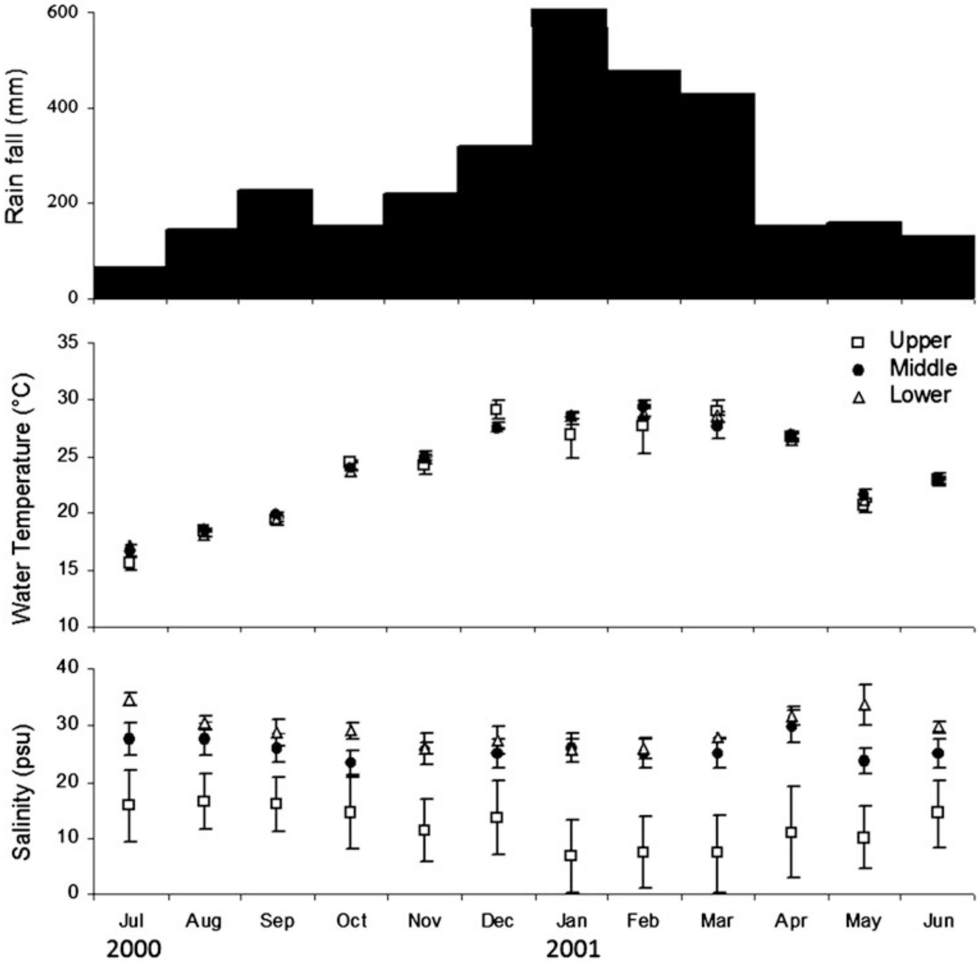
resulting in an environmentally stochastic stress zone (Attrill and Rundle, 2002). They have been well studied in terrestrial systems and at terrestrial/aquatic interfaces, including the interface between rivers and estuaries.

Ecoclines have been only recently defined as gradient zones containing relatively heterogeneous faunal communities, which are environmentally more stable than those of ecotones (Attrill and Rundle, 2002). Ecoclines represent a boundary of more gradual and progressive change between two systems, resulting from a response to stepwise differences of a major environmental driver that controls the transition process and is followed or modified by other forcings.

The environmental gradients occurring in estuaries can be either classified as ecotones, represented by the transition community between the freshwater and marine species, or ecoclines comprised of a series of assemblages along the estuarine salinity gradient (Attrill and Rundle, 2002). Other potential environmental influences on estuarine ecoclines would be the seasonal changes in freshwater flow (long-term process) or daily rhythm of the tides (short-term process) that have a strong influence on the estuarine community.

Consequences of man-driven changes in environmental gradients

Estuaries have been historically impacted by anthropogenic activities, such as land reclamation, functioning of ports and maritime terminals, harboring of industrial estates, and dredging of waterways, all of which are part of a non-sustainable economic model which has failed to take into consideration any ecological limits of the estuarine environment (Barletta et al., 2010). The effects of frequent and intense dredging activities in estuaries, for example, can drastically alter the physical-chemical conditions of the system and change the patterns of their environmental gradients. For example, comparisons of



Environmental Gradients, Figure 3 Total rainfall (mm) and mean (±sd) values of water temperature (°C) and salinity in Paranaguá Estuary (upper, middle, and lower estuary).

demersal fish communities in the main channel of the Paranaguá estuary (South Brazil – tropical and subtropical transition zone) before, during, and after dredging (Barletta et al., 2008, 2010) showed evidence of impacts. The authors concluded that, in this estuary, during dredging, significant differences were observed in fish species, species composition (biodiversity), *Genidens genidens* (Cuvier), and *C. spixii* (density and biomass). Most of these species had the highest densities and biomasses in the main channel of the estuary during the dredging process. However, *Cynoscion leiarchus* (Cuvier) and *Menticirrhus americanus* (L.) had the lowest density and biomass values during and after the dredging process. These results strongly suggest that dredging had an influence on the fish assemblage composition and its distribution in the estuary.

Summary

Environmental gradients in estuaries are directly linked to the seasonal and diel fluctuations of freshwater discharge from rivers and tidal action that influence abiotic and biotic changes and generate ecoclines (Barletta et al., 2008). These abiotic parameters affect the biotic communities inhabiting estuaries that must adapt to the variable environmental conditions. Fishes, for example, must have a strong eurythermohaline capacity to tolerate frequent fluctuations of salinity and temperature. Moreover, some anthropogenic factors that can cause changes in estuarine gradients, such as dredging of waterways and removal of riparian vegetation, result in significant sediment and nutrient inputs that can impact the systems (Barletta et al., 2010).

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Cross-references

Anthropogenic Impacts
Dredging
Fish Assemblages
Tidal and Nontidal Oscillations
Tidal Hydrodynamics
Tides

EPIBIONT

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Synonyms

Epifauna; Epiflora

Definition

An epibiont is an organism living on the surface of another living organism. The relationship between the two organisms may be neutralistic or commensalistic. The host of the epibiont is referred as the basibiont.

Epibiosis process

Epibiosis is a direct consequence of surface limitation and results in spatially close associations between two or more living organisms belonging to the same or different species. These associations can be specifically guided by host chemistry resulting in species-specific symbiotic or pathogenic assemblages. Most colonizers, however, are nonspecific substratum generalists, and epibionts are able to survive in the natural environment longer than free-living forms and, by means of adhesive strategies, they can adapt to adverse conditions; e.g., organic matter limitation, and symbiosis of microorganisms with macroorganisms is a widespread phenomenon that should have a profound impact on the physiology, ecology, and evolution of both hosts and symbiotic partners (Laudien and Wahl, 2004).

In epibiosis, the ecological consequences for the overgrown host (basibiont) and the colonizer (epibiont) can be positive and negative. The distribution of the epibiont species is related to specific basibiont species and the pattern of colonization of the epibiont community (Fernandez-Leborans and Gabilondo, 2006).

Epibiont populations should have relevant functions and roles at the ecosystem level; thus, epibionts with high densities are involved in energy transfer to higher trophic levels. In addition, the colonization of a marine hard-bottom community on newly available substrata is governed by presettlement (survival and distribution of colonizing stages), settlement (composition of colonizer pool, competence of settling stages, substratum preferences), and post-settlement processes (competition, consumption, etc.). When the substratum becomes the limiting factor during recruitment, dominant competitors should lead competitively inferior species to extinction (Laudien and Wahl, 2004).

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Cross-references

Benthic Ecology

ESTUARINE BEACHES

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Synonyms

Littoral drift; Littoral transport; Longshore transport

Definition

Estuarine Beaches. Estuarine beaches are unvegetated or partially vegetated sand, gravel, or shell intertidal beaches in a partially enclosed bay, lagoon, or fjord and similar features connected to oceans or seas (Nordstrom, 1992).

Introduction

Estuarine beaches include unvegetated or partially vegetated sand, gravel, or shell intertidal beaches in a partially enclosed bay, lagoon, or fjords and similar features connected to oceans or seas (Nordstrom, 1992). They are created by locally generated waves and wave-induced currents along estuarine shorelines around the world in a variety of coastal settings, fetch exposures, and tide ranges. Fetch is the distance over open water that wind can blow and generate surface waves. World-wide, many types of bays with beaches occur; this document is intended to describe short-fetch, intertidal beaches not typically found on lakes or on the ocean. While similarities certainly exist, this document does not describe shorelines that may be sheltered from the ocean but are still high energy or shorelines that are relatively low energy but have long fetches.

While estuarine beaches are similar to their ocean counterparts in recreational and ecosystem services, they tend to be smaller in dimension. Ecosystem services are the benefits resulting from the resources and processes that are supplied by ecosystems. The function and dimension of a beach are influenced by wave action, and the physical type of sediment will vary depending on the geologic setting. Beaches can extend shoreward from the location of significant wave reworking below the low tide line to the landward limit of wave reworking during storms. In developed estuarine environments where beaches may owe their origin, dimensions, and some of their surface characteristics to human action, this definition is still valid because the beaches are freely worked by waves and wave-induced currents (Nordstrom, 1992).

Physical processes

A sufficient volume of sediment is necessary for beach development. The sediment is derived from eroding fastland banks or from fluvial flow. Bank sediments typically consist of fine-grained silt and clay as well as sand and gravel. Sand is defined as grains sized between 0.1 and 2.0 mm (Folk, 1980). As a bank erodes, the material slumps to the base, and wave action winnows out the finer sediments and reshapes the coarser material into a beach (Hardaway and Byrne, 1999). While most of the material on a beach tends to be sand-sized particles, coarser sediments consisting of shells, gravels, pebbles, or even cobbles can be deposited as a beach feature on the foreshore. The sand may be predominantly terrigenous quartz in mid- to upper latitude settings and calcium carbonate in more tropical settings. Calcium carbonate beaches are sourced by reworking of offshore coral reefs.

In a typical small estuary, beaches can exist adjacent to the fastland banks and/or marsh shorelines (Figure 1: Top). When erosion-resistant fastland banks occur, pocket beaches may form between headlands. Pocket beaches may also form between coastal structures (Figure 1: Middle). Because sand can be transported through the inlet, beaches can occur just inside the estuary. Superposed beaches, sandy material above a vegetated intertidal zone, can occur in areas with larger tide ranges. Shoreline orientations can differ greatly over short distances, causing great differences in exposure to dominant winds which will isolate beach segments and limit longshore sediment exchanges. This will result in beach sediments that closely resemble local source materials (Nordstrom and Jackson, 2012).

Estuarine beaches can occur along shorelines with fetch exposures from less than 100 m in small creeks to the length of the estuary. While most exist along fetch-limited, relatively low wave energy coasts, exceptions may occur along more open reaches of a bay or lagoon which experience larger waves due to their proximity to the open ocean. Along fetch-limited coasts, fetch distance, rather than wind duration, limits wave generation. Ocean waves can enter through an inlet and influence shorelines many kilometers into an estuary; however, ocean waves tend to lose much of their energy through refraction or breaking, and their influence decreases the farther they travel into the estuary (Nordstrom, 1992). In fetch-limited environments, wave heights generated by local winds depend principally on wind conditions (speed, duration, and direction) and

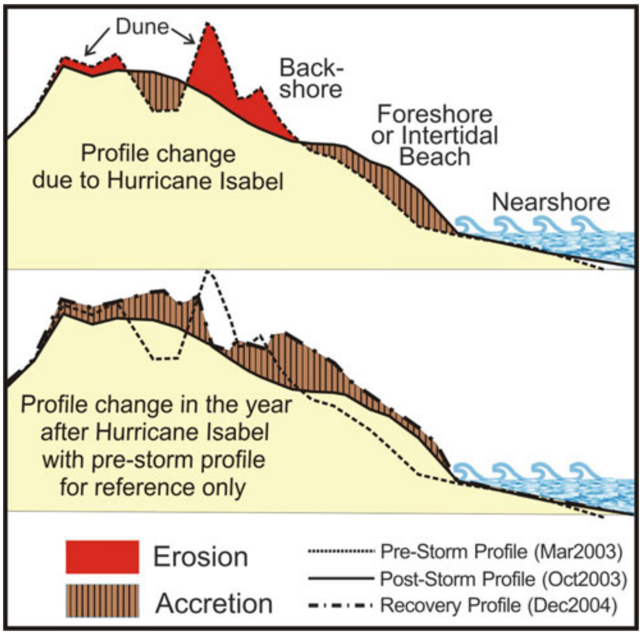
Estuarine Beaches, Figure 1 Photos of estuarine beaches in Chesapeake Bay, USA. *Top*: Estuarine beaches occupy many different geomorphic environments including in front of marshes. *Middle*: A pocket beach has developed in front of an eroding upland bank that is between two hardened headlands. *Bottom*: A long, linear beach that has a narrow, sand backshore which transitions to forest.

water body dimensions (width, length, and depth) (Jackson et al., 2002). In estuaries with small basin dimensions, the energy potential of waves is limited, resulting in beach characteristics that can include longshore bars in the nearshore, narrow foreshores, narrow or nonexistent backshores, short beach lengths, low tide terrace, and poorly developed or nonexistent dunes (Nordstrom and Jackson, 2012). In addition, ship and boat wakes can increase erosion along estuarine shorelines.

Tidal and surge-related water-level fluctuations affect low wave energy beach morphology and processes. Tide range affects the strength of tidal currents and the vertical distribution of wave energy over the profile which, in turn, determines the width of the beach and the duration that the breaking waves will occur at any elevation (Nordstrom and Jackson, 2012). Tidal currents can be the mechanism for sediment transport in areas where waves may not break. Increased water levels during storms increase the distance inland the larger storm waves will travel which has a greater effect on the shoreline. Storm surge can vary greatly throughout an estuary depending on tide range, wind direction, and volume of water associated with the event. This, along with shore orientation, will determine a storm's effect on a particular stretch of shoreline.

Estuarine beaches will develop in a variety of geomorphic settings from small pocket beaches to longer linear features to spits (Varnell et al., 2010). Many beaches exist in front of marshes and creek mouths in addition to upland banks (Figure 1: Top). As noted, wave climate, tide range, sediment source, shore orientation, and shore morphology all help determine the dimensions of an estuarine beach system, and configurations vary greatly. Some beaches exist as sand only in the intertidal area of a coast (Figure 1: Bottom), while others have well-developed nearshore bars and dune systems.

Estuarine beaches are often part of a larger coastal profile that includes dunes on the landward side and sand bars in the nearshore. If sufficient sand is available and the beach has a backshore wide enough to support dune vegetation, aeolian processes can create a dune adjacent to the beach. Beaches and dunes also act as store houses of sand during storm events where the beach profile adjusts to the larger impinging wind wave climate and higher water levels by depositing sand in the nearshore or other areas of the beach cross-sectional profile. During the post-storm period, calmer waves can move the sand back onshore allowing the beach and dune to recover (Figure 2).



Estuarine Beaches, Figure 2 Cross-sectional beach profile data showing erosion and accretion of the beach and dune associated with Hurricane Isabel and the accretion on the entire beach during the post-storm recovery period (Modified from Hardaway et al., 2009).

Ecosystem services

Estuarine beaches and dunes provide a setting for a variety of ecosystem services. These include but are not limited to habitat, nutrient uptake, food production, wave attenuation, sediment stabilization, and recreation (NRC, 2007). These sheltered coastal environments provide habitat for a variety of organisms. Large populations of horseshoe crabs occupy the beaches of Delaware Bay to lay eggs (Brady and Scharding, 1998). The shallow refuge areas of shoreface environments provide suitable conditions for pupping some shark species. Some turtle species nest on upper beaches and within low dune areas in bays and estuaries. Estuarine beaches are home to endangered or threatened species.

Natural beach litter in wrack lines on the foreshore or backshore tends to be more prevalent on low-energy beaches due to the large amounts of vegetation growing in sheltered waters and the numerous breaks in shore orientation. This material can have a pronounced effect on beach topography (Jackson et al., 2002) and dune accretion (Milligan et al., 2005). The beach detritus also provides habitat and food for meiofauna.

Management

Humans rely on estuaries for food, recreation, and jobs, and our activities have led to a decline in the health of estuaries, making them one of the most threatened ecosystems on Earth. Of the 32 largest cities in the world, 22 are located on estuaries (Ross, 1995). In many estuaries throughout the world, beaches have been eliminated because of the building of seawalls, bulkheads, and revetments for erosion control and harbor creation (Figure 3: Top). This hardening of the shoreline can reduce or eliminate erosion of material from the upland which would become part of the longshore transport system.

Human influence on estuarine beaches also can be positive. Other shore erosion control structures, such as groins, jetties, and breakwaters, can be littoral barriers which allow

Estuarine Beaches, Figure 3 *Top*: Bulkheads in low-energy estuaries occupy the intertidal zone instead of beach or marsh habitat. Often, no buffer exists. *Bottom*: breakwater and beach fill create a wide, protective recreational beach.

sand to accumulate over time, increasing beach width which leads to dune creation. However, in some cases, they can also be a mechanism for beach loss since shores downdrift may have accelerated erosion rates. Care must be taken in the design of shore structures. The purposeful addition of sand, called beach nourishment, can enhance existing sandy shorelines or create beaches where one previously existed, but because of erosion, development, or change in longshore transport, it no longer does. Structures such as breakwaters are used to hold the beach nourishment in place (Figure 3: Bottom). These nourished beaches can provide the same ecosystem services as natural habitats. Fenster et al. (2006) found that the threatened Northeastern beach tiger beetle, *Cicindela dorsalis dorsalis*, rapidly moved into renourished beach areas.

Summary

Estuarine beaches have both ecological and recreational value. Due to ongoing coastal development, many estuarine beaches are impacted both negatively and positively. As eroding sand shorelines are hardened by bulkheads and revetments, a source of sand may be removed from the littoral reach system resulting in beach loss. Beach nourishment may offset some of these losses. As is often the case, the local, state, and federal permitting may allow or disallow certain types of coastal construction. It is clear that estuarine beaches provide quality habitat for a variety of species including man. Therefore, thoughtful management is a key to ensuring estuarine beach longevity in the coastal environment.

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Cross-references

Beach Management
Beach Processes
Estuarine Geomorphology
Foredune
Shoreline
Wrack

ESTUARINE CIRCULATION

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Definition

An estuary is a semi-enclosed coastal body of water which has a free connection with the open sea and within which salt water is measurably diluted with fresh water derived from land drainage.

Introduction

Estuaries are unique aquatic environments that share terrestrial and marine contributions, creating a set of interesting physical characteristics including bimodal circulation patterns (fresh water flowing along the surface and saline water entering along the bottom) and mixing processes induced by tides. To a lesser extent, continental shelf processes such as storm surges and wind-driven circulation can also influence circulation in estuaries. At first glance, most estuaries are governed by tidal oscillation and influenced by river discharges, so mixing processes account for most of the hydrodynamic characterization of estuaries. The resulting velocity and salinity fields have strong time variability, and the non-steady-state current patterns may be altered by sediment erosion, transport, and sedimentation, which contribute to bathymetric and margin changes.

Formation and geological age

Estuaries are transitional environments that were formed in the narrow coastal boundary zone between the ocean and the continents during the Holocene rise in sea level which began about 15,000 years ago, when the sea was about 125 m below its present level. They were formed due to the Flandrian Transgression, the secular increase of global sea level that ended about 7,000 years ago. Local changes in sea level as well as tectonic activity may have also contributed to the formation of estuaries. At the end of that period, the continental coastal plains and the river valleys were slowly flooded forming the bays, coastal lagoons, inlets, and estuaries.

The geomorphology of estuaries depends on coastal topography, geomorphology, continental shelf characteristics, and river discharges, which change due to natural processes of erosion and sediment deposition. These processes have often been modified due to the anthropogenic activities in estuaries and in their drainage basins.

Recent studies conclude that during the late Holocene sea-level rise amounted to 5–7 m. This was corroborated by the earlier work of Fairbridge (1961). Villwock (1972), Suguio and Martin (1978), and Kowsmann et al. (1977) also confirmed this phenomenon in the south and southeastern coastal plain of Brazil.

Transitional environments associated with estuaries, such as salt marshes and mangroves, are colonized by the salt-tolerant species and distributed in regions of temperate and tropical climates. They are environments ultimately dependent on the dynamics of estuarine systems, but have been strongly affected by human activities around the world Herz (1992) . As such, their areas have declined significantly. Some have been protected by reserve status since the nineteenth century. Human impacts and their interference on estuaries and coasts are described in French (1997).

Herz (1992) used digital mapping, based on processing of multispectral images, to describe the mangrove colonization area along the 8,000 km of the Brazilian coast. The mangrove area decreased from 85.0 % in the northern coast of Brazil to 9.9 % and 5.1 % in the eastern and southern coastal regions.

Pioneer investigations

In 1875, the Swedish researcher F. L. Ekman performed experiments in the Götaelf River Estuary (Sweden) to investigate estuarine processes (quoted in Defant (1961), 539). From the analysis of the salinity distribution on August 1875, Ekman observed that a strong outflowing current could not be compensated only by river discharges, but instead by a compensating upward motion (upwelling) of saline water from the bottom.

Upwelling is a vertical advective motion that breaks the continuity of the bidirectional motion field observed by Ekman. The vertical motion was calculated theoretically under steady-state conditions for the first time by Pritchard (1954), solving the salt balance equation using extensive experimental data from the James River Estuary (Virginia, USA) during the summer of 1950.

Ekman investigated the occurrence of currents up-estuary in the lower layers of the estuary in a paper published in 1899, quoted in Defant (1961, 539). Although this paper was only descriptive, Ekman was aware that the river discharge input and the salinity stratification were the main forcing mechanisms of the observational results.

It was only in 1952 – almost 60 years later – that D. V. Pritchard became the first researcher to link estuarine circulation to the forcing by the horizontal density gradient. He used observations from the James River estuary to demonstrate this mechanism, quoted in Geyer (2010, 13).

Definitions and variability

Ketchum (1951) focused on the exchanges of fresh and salt water in tidal estuaries and defined an estuary as “a body of water in which the river water mixes and measurably dilutes sea water.” The most classical definition is that stated by Pritchard (1952) and Cameron and Pritchard (1963), "An estuary is a semi-enclosed coastal body of water which has a free connection with the open sea and within which salt water is measurably diluted with fresh water derived from land drainage." Several other definitions are found in the literature. Perillo (1995) defined an estuary as "a semi-enclosed coastal body of water that extends to the effective limit of tidal influence, within which sea water entering from one or more free connections with the open sea, or any other saline coastal body of water, is significantly diluted with fresh water derived from land drainage, and can sustain euryhaline biological species from either part or the whole of their life cycle." These definitions along with others by Dionne (1963), Kjerfve (1987), and Dalrymple et al. (1992) have the ability to include all basic marine disciplines, which are necessary for studying this transitional water body as an ecosystem. There are interchanges between three estuarine domains: the tidal river (TR), the mixing zone (MZ), and the estuarine plume (EP), which must be investigated together as a unique transitional system Okubo (1970).

In these definitions, salt-water dilution is due to mixing processes, which involve advection (large-scale) and diffusion (small-scale). These occur internally in the estuary and tend to produce uniformities in the concentration of dissolved properties (Bowden, 1963; Okubo, 1970). The small-scale component of mixing is generated by surface (due to the wind stress) and bottom shears and the vertical internal shears.

The estuarine drainage basin and the fresh water discharge (Q_f) into the estuary may change as a result of natural processes and human activity, such as inadequately planned land use and pollutants from homes, farms, and factories. According to Ji (2008), every surface water system is unique and many face similar environmental problems such as eutrophication, pathogen contamination, toxic chemicals, loss of habitat, and declines in fish and wildlife. Estuaries also have problems that cause declines in water quality, living resources, and overall ecosystem health. Rivers, lakes, and estuaries contain a very small fraction of the total earth's total water budget driven by the hydrologic cycle.

Circulation variability in estuaries is classified as intertidal when it occurs at semi-diurnal or diurnal tidal frequencies (>1 cycle/day), or subtidal at lower frequencies (<1 cycle/day). The beat period of the main semi-diurnal tidal lunar and solar components (M_2 and S_2) results in fortnightly subtidal frequency (modulation between successive spring tides, ≈15 days).

Classification of estuaries

Advection and mixing processes can be used to compare and classify different estuaries. The major classification schemes are based on salinity stratification, circulation, and mixing. Figure 1 shows a simple steady-state dynamic balance analysis of the interaction of these processes based on a longitudinal section presented by Geyer (2010).

The first estuary classification was suggested by Stommel (1951) taking into account the main forces as tides, fresh water discharge, and wind. Raritan, Pamlico Sound, and Mississippi river estuaries of New Jersey, North Carolina, and Louisiana (USA) have as primary forcings the tide, wind, and river, respectively. The important characteristic of these estuaries is the vertical salinity stratification. The Raritan River estuary, which is the shallowest one, is nearly vertically homogeneous. The Mississippi River estuary is the deepest, being forced by micro-tides.

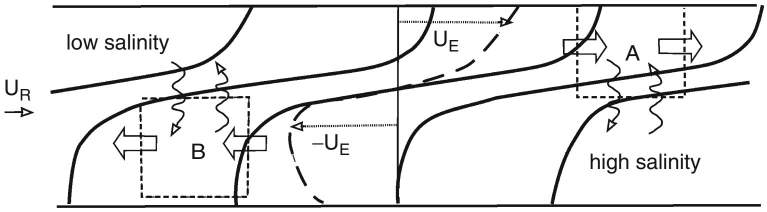
Pritchard (1952) introduced a classification scheme based on the main geomorphologic estuarine features. Four types of estuaries were differentiated: (1) coastal plain estuaries or drowned river valleys; (2) bar built estuaries; (3) fjords, typically in higher latitude regions; and (4) tectonic estuaries.

Taking into account vertical salinity stratification, Stommel (1953) suggested the following classification: (1) salt wedge estuaries, river discharge dominates with vertical mixing absent; (2) fjords, deep estuaries (several hundred meters) characterized by a highly stratified upper layer; and (3) moderately and highly stratified estuaries, dependent on the intensity of the vertical mixing and establishment of a steady-state bidirectional circulation. This classification was described by Pritchard (1955), Cameron and Pritchard (1963), and Schubel and Pritchard (1972), taking into account improvements related to dynamic conditions due to estuarine circulation and salinity stratification.

During the 1960s, estuarine classifications were based on quantitative criteria. Ippen and Harleman (1961) introduced the stratification number, based on laboratory experimental results, and Hansen and Rattray (1966) introduced the stratification and circulation parameters.

Hansen and Rattray (1965) were the first investigators to link estuarine classification with its physical properties. They used salinity stratification and circulation in a theoretical development based on the analytical solutions of a bi-dimensional system of equations (motion, mass and salt conservation, and a linear state equation) under steady-state conditions, applied to a laterally homogeneous estuary. The results were vertical profiles of the longitudinal velocity and salinity used for a theoretical deduction of the stratification-circulation diagram (Hansen and Rattray, 1966). The coordinates (axis) of this diagram are the stratification (p_e) and the circulation (p_c) parameters defined by $p_e = \frac{S_f - S_s}{\overline{S}} = \frac{\delta S}{\overline{S}}$ and $p_c = \frac{u_s}{u_f}$, respectively, where S_f and S_s are the time mean salinities on the bottom and surface, respectively, and $\overline{S}$ its mean-depth value. The velocity u_s is the time mean value on the estuary surface and u_f is the one generated by the river discharge (Q_f) defined by $\left(u_f = \frac{Q_f}{A}\right)$, where A is the cross-sectional area.

The stratification (p_e) and circulation (p_c) parameters are linked to a third parameter $\underline{\upsilon}$ defined by $\upsilon = \frac{\phi_D}{\phi_D + \phi_A}$,



Estuarine Circulation, Figure 1 Schematic longitudinal section of an estuary showing the influence of advection (⇒due to river discharge U_R and tidal currents) and vertical mixing (thin waving lines) in the local salt balance. The thick lines are isohalines. Box A (*upper layer*), horizontal advection causes a reduction of salinity, but vertical mixing compensates by replacing the low-salinity water with underlying high-salinity water. The relative roles of advection and mixing are reversed in Box B (*lower layer*). Bidirectional circulation is shown in the vertical profile of the u-velocity component (*dashed line*) with the depth of no motion, and U_E and $-U_E$ are upper and lower maxima values shown in this profile. According to Geyer (2010).

where ϕ_D and ϕ_A are the salt flux into the estuary due to turbulent diffusion and the advection, respectively. When $\phi_D >> \phi_A$, it follows that $\upsilon \to 1$ and the salt flux are dominated by the tidal mixing; for $\phi_D << \phi_A$, $\upsilon \to 0$ and the salt flux is dominated by advection (gravitational circulation). Then, this parameter varies from 0 to 1, and $\upsilon = \upsilon(p_c, p_e)$ is expressed theoretically by Miranda et al. (2012):

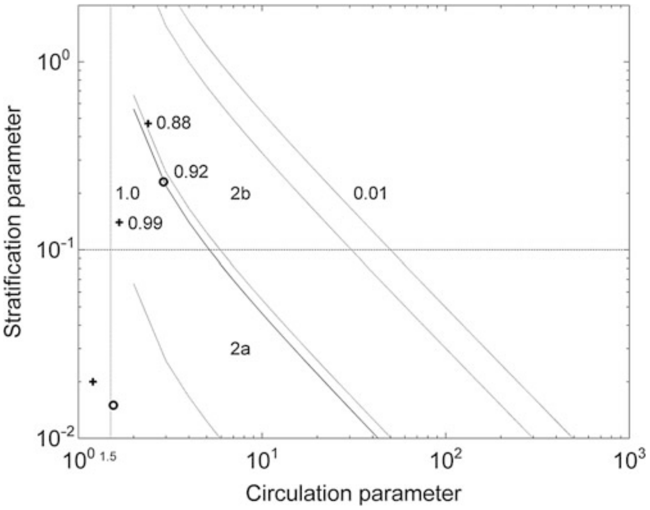
$$(p_e)^{-1}[210 + 252(p_c - 1.5)]\upsilon^2 + \left[32 - (p_e)^{-1}(210 + 252(p_c - 1.5)) + 76(p_c - 1.5) + \frac{152}{3}(p_c - 1.5)^2\right]\upsilon = 0, \tag{1}$$

For $\upsilon = 0$ (1) has no physical meaning, and for $\upsilon = 1$ the salt transport is due to the turbulent diffusion only. The equation is thus reduced to:

$$32 + 76(p_c - 1.5) + \frac{152}{3}(p_c - 1.5)^2 = 0 \tag{2}$$

This equation with the unknown $(p_c - 1.5)$ has no solution in the field, unless the constant 32 is disregarded. If so, it has two solutions: $p_c = 0$ (with no physical meaning) and $p_c = 1.5$, which indicates that when the salt flux is due to the turbulent diffusion ($\upsilon = 1$), its solution is independent of the stratification parameter (p_e). Under this simplification, it is possible (1) to define a set of isolines in a Cartesian Coordinate system ($p_e \times p_c$) with $\upsilon = cte$ and interval $0 < \upsilon \leq 1$.

Hansen and Rattray (1966) confirmed that theory by analyzing an experimental data set from several estuaries, from which four previously classified estuarine types emerged: (1) Type 1, well-mixed estuary (unidirectional circulation); (2) Type 2, partially mixed estuary (circulation reverses at depth); (3) Type 3, fjords; and (4) Type 4, salt-wedge estuary. Types 3 and 4 were classified because their experimental data fits very well in the stratification-circulation diagram. Subdivisions a and b for Types 1, 2, and 3 are low and high stratification when $p_e < 0.1$ and $p_e > 0.1$, respectively.



Estuarine Circulation, Figure 2 Stratification-circulation diagram with experimental data of the Caravelas River Estuary (southern Bahia State, Brazil) for the August (spring and neap tide) and for January (spring tide) experiments. The values of the parameter υ are indicated close to the symbols + and o. The circulation parameter (u_s/u_f) was approximate to u_s/u_a, were u_a is the residual velocity (time mean-depth value).

Figure 2 shows application of this diagram, with observational data for the tropical Caravelas River Estuary, located in the SE Brazilian coast (lat. 17°45′14.0″; long. 039°13′53,0W″). The classification changes from well mixed ($\upsilon \approx 1.00$ – all salt transport is due to diffusion in the spring tide) to partially mixed with low stratification ($\upsilon = 0.80$, meaning that 80 % and 20 % are the up-estuary salt transport due to diffusion and advection, respectively, at neap tide), due to the fortnightly tidal modulation.

These theoretical results were revisited and confirmed with the introduction of alternative parameters by Fisher (1972), Prandle (1985), Jay and Smith (1988), and Scott (1993). In Prandle's paper, the nondimensional p_c axis was replaced by the ratio of the residual accelerations associated with the horizontal density gradient and bed friction, yielding a more direct assessment of the classification based on more readily available parameters. The demarcation line, which separates estuaries of types 1 and 2, can then be explained by the occurrence of flow reversal. Prandle's diagram was applied by Miranda et al. (2012) to classify the estuarine Bertioga Channel (São Paulo).

Two classification diagrams were recently developed in estuarine physics. They introduced parameters based on salinity stratification and estuary circulation: the vertical Ekman (E_K) and the Kelvin (K_e) numbers (Valle-Levinson, 2008) and the nondimensional tidal (U_T) and freshwater (U_R) velocities (Geyer, 2010).

The Amazon River is the largest river system on Earth, containing ~20 % of the global fresh water supply. The large Amazonian drainage basin, which exceeds 7×10^6 km^2, combined with the equatorial and tropical climate results in many tributaries and tremendous discharge. Tidal ranges are as high as 6 m at the mouth of the Amazon, where intertidal and subtidal periodicities are the dominant control on river-level changes, and these tidal influences extend more than 800 km upstream to Óbidos (Sioli, 1984; Oltman, 1967; quoted in Archer, 2005, p. 18). Many aspects of this transitional system are unique and not easily characterized within the existing definitions and classifications. The extreme tidal oscillations at the mouth create ideal conditions for the development of tidal bores throughout the mouth and inner areas, as first described by Rongel (1943). Kjerfve and Ferreira (1993) made time series measurements of water level, velocity, salinity, and temperature in the presence of a tidal bore in the macrotidal Mearin River (São Marcos Bay, Maranhão) in northeastern Brazil. This hydrodynamics was complex with an ephemeral flow of 1.5–2.0 ms^{-1} transient velocity surge and propagation speed as high as 7.2 ms^{-1}.

The estuarine coastal embayment of the Amazon mouth (enclosing the North and South channels) is nearly 300 km in width. These channels are not estuaries sensu stricto in terms of saltwater-freshwater mixing and dilution (Bowden, 1978; quoted in Archer, 2005). Thus, for the Amazon system the definition of a drowned river valley estuary with an inner delta can be applied until further investigations are carried out.

Equations of motion, mass and salt conservation

Oceanic tides and land runoff are typical examples of processes that control the hydrodynamics in estuarine environments. Advection and small-scale turbulent motions affect salinity and temperature mixing processes, among other physical aspects related to erosion and transport of pollutants and organisms.

The basic system of equations that drives estuarine circulation and mixing are the mass and salt conservation equations, the momentum conservation, and the equation of state. Fluid density (ρ) and its velocity $\left(\vec{v} = u\vec{i} + v\vec{j} + w\vec{k}\right)$ will be assumed continuous functions of space and time in a Cartesian Coordinate System (Oxy is the horizontal plane and Oz the depth, oriented against the gravity acceleration, $\vec{g}$). With Ox oriented along the estuary axis, the velocity components u and v are named longitudinal and transversal (or secondary), respectively.

The relationship between fluid density (ρ) and its velocity ($\vec{v}$) is derived from the principle of mass conservation (continuity equation). Its analytical deduction may be made with different theoretical developments found in oceanography texts (Sverdrup et al., 1942; Lacombe, 1965; Kinsman, 1965, among others). A convenient mathematical expression is the Eulerian formulation:

$$\frac{\partial \rho}{\partial t} + \nabla \bullet \rho \cdot \vec{v} = 0, \text{ or } \frac{1}{\rho}\frac{d\rho}{dt} + \nabla \bullet \vec{v} = 0 \tag{3}$$

The symbol • indicates the scalar product of the gradient operator ∇ by the mass flux vector $\rho \cdot \vec{v}$ $\left(\left[\rho \cdot \vec{v} = ML^{-2}T^{-1}\right]\right)$. It states that the local (total) variation of the density is in balance with the divergence of the mass flux vector. If the fluid density is constant (ρ = cte), or the density does not change during the motion $\left(\frac{d\rho}{dt} = 0\right)$, the fluid is defined as incompressible and the continuity equation simplifies to:

$$\nabla \bullet \vec{v} = 0, \text{ or } \frac{\partial u}{\partial x} + \frac{\partial v}{\partial y} + \frac{\partial w}{\partial z} = 0 \tag{4}$$

The continuity equation holds for a laminar single fluid flow like water. However, if we approximate the estuarine water mass as oceanic seawater, which is a binary fluid (pure water + salt) and usually in turbulent motion, we must be aware of the following approximations: (1) the velocity $\overline{\vec{v}}$ is the time mean value of the turbulent velocity (the over bar indicates a time mean value); (2) the mass conservation is also valid for the turbulent velocity, $\nabla \bullet \vec{v}_{turb} = 0$, with $\vec{v}_{turb} = \vec{v} - \overline{\vec{v}}$; and (3) the net salt diffusion across a closed boundary may be disregarded, which is a good approximation as demonstrated by Csanady (1982).

When (4) is applied to an estuary and the details of its circulation are not known in the interior fluid domain, we may use the continuity equation in its integrated form, using the Gauss theorem (or divergence theorem), under the assumption that all geometric and physical properties have all the regularity conditions imposed by its hypothesis. Then, if V denotes the estuarine volume boundary, A is a closed area and $\vec{n}$ is its normal unity vector oriented positively from the interior to the exterior, then:

$$\int_V \left(\nabla \bullet \vec{v}\right) dV = \int_A \left(\vec{v} \bullet \vec{n}\right) dA = 0 \tag{5}$$

the surface integral, which is the volume transport (L^3T^{-1}, m^3 s^{-1}) through its closed boundary is zero. In estuaries, the closed area A has free and bottom boundaries which may have mass contributions from external water sources or sinks (evaporation, precipitation, and subsurface water springs from aquifers), and two vertical sections enclosing the estuarine water body.

A simple integrated equation of salt conservation, taking into account only the advection process on salt mixing, may be easily obtained applying the divergence operator to the mass flux vector $S\cdot\rho\,\vec{v}$ $\left[S\cdot\rho\,\vec{v}\right] = M\cdot L^{-2}T^{-1}$. Under the hypothesis that the salinity (density) and the velocity field in the estuarine domain are in steady state represented by its time mean value during tidal cycles, the divergence of the salt mass flux is zero:

$$\nabla\bullet S\rho\,\vec{v} = 0 \tag{6}$$

and as in (5)

$$\int_V\left(\nabla\bullet S\rho\,\vec{v}\right)dV = \int_A\left(S\rho\,\vec{v}\bullet\,\vec{n}\right)dA = 0 \tag{7}$$

the salt transport (MT^{-1}, kg s^{-1}) over a closed area is zero.

With simplified geometry, (5) and (7) may be transformed in a system with two unknowns. They were used in the Knudsen hydrographic theorem applied for the first time around 1,900, enabling mean velocities to be calculated in cross-section areas under steady-state conditions, with known mean salinities in a highly stratified estuary (salt wedge) and with the fresh water discharge (Q_f) as the main forcing mechanism. In this type of estuary, the salt transport is driven by river discharge and the vertical turbulent mixing is absent. Let A_s (u_s) and $A_i(u_i)$ be the superior and inferior cross-section areas (mean velocities) limited by the halocline, and S_s and S_i the mean salinities, respectively.

Disregarding the mass inflow and outflow across the free surface and the bottom, the equation system of (5) and (7) may be applied, taking into account that $\vec{v}\bullet\vec{n}\neq 0$ only on the areas A_s and A_i, then:

$$\int_A\left(\vec{v}\bullet\,\vec{n}\right)dA = u_sA_s - u_iA_i - Q_f = 0, \tag{8a}$$

$$\int_A\left(S\rho\,\vec{v}\bullet\,\vec{n}\right)dA = S_s\rho_s v_s A_s - S_i\rho_i v_i A_i = 0 \tag{8b}$$

Disregarding the density differences in the upper and lower layers ($\rho_s \approx \rho_i$) in the (8b), this system may be resolved for the mean current velocities (u_s, u_i) and/or the volume transports (Q_s, Q_i):

$$u_s = \frac{S_iQ_f}{A_s(S_i - S_s)}, \quad \text{or} \quad Q_s = \frac{S_iQ_f}{(S_i - S_s)}, \tag{9a}$$

$$u_i = \frac{S_iQ_f}{A_i(S_i - S_s)}, \quad \text{or} \quad Q_i = \frac{S_sQ_f}{(S_i - S_s)}. \tag{9b}$$

An application of this result may be found in Miranda et al. (2012) using the following experimental data of the Fraser River, according to Geyer (1986): discharge $Q_f = 3{,}000\ m^3s^{-1}$; sections geometry $A_s = 3{,}750\ m^2$ and $A_i = 4{,}500\ m^2$, and salinities $S_s = 14.0$ psu and $S_i = 30.0$ psu. Then, the theoretical mean velocities and volume transports are $u_s = 1.5\ ms^{-1}$, $u_i = 0.6\ ms^{-1}$, $Q_s = 5{,}525\ m^3s^{-1}$, and $Q_i = -2{,}525\ m^3s^{-1}$, respectively. These results show that the transport volumes are in balance with the fresh water discharge.

Consider an estuary with a surface area (A) delimited by the bottom and the surface and two vertical sections A_1 and A_2 at the river zone (RZ, where $S = 0$) and mixing zone (MZ), respectively. It follows from (8a), the mean longitudinal velocity under steady-state conditions across the area A_2, which is given by: $-u_f\cdot A_1 + u_2\cdot A_2 = -Q_f + u_2\cdot A_2 = 0$, and thus $u_2 = u_f = Q_f/A_2$ is the fresh water velocity driven by the river discharge (Q_f).

From the continuity equation (3), the conservation salt principle due only to the advective process is:

$$\frac{\partial(\rho S)}{\partial t} + \nabla\bullet\rho S\,\vec{v} = 0 \tag{10a}$$

and

$$\frac{\partial S}{\partial t} + \vec{v}\bullet\nabla S = 0 \tag{10b}$$

However, the local variation $\left(\frac{\partial S}{\partial t}\right)$ also depends on the turbulent salt-diffusion flux (ϕ_S), which is simulated by Fick's law $\left[\phi_S = -D\left(\frac{\partial S}{\partial n}\right)\right]$ (D is the dynamic diffusion coefficient ($[D] = M{\cdot}L^{-2}\ T^{-1}$) and $\frac{\partial S}{\partial n}$ is the directional salinity gradient). The composition of this partial salt flux with (10b) takes the expression of the salt conservation equation (Sverdrup et al., 1942; Pritchard, 1958):

$$\frac{\partial S}{\partial t} + u\frac{\partial S}{\partial x} + v\frac{\partial S}{\partial y} + w\frac{\partial S}{\partial z} = \frac{\partial}{\partial x}\left(K_x\frac{\partial S}{\partial x}\right) + \frac{\partial}{\partial y}\left(K_y\frac{\partial S}{\partial y}\right) + \frac{\partial}{\partial z}\left(K_z\frac{\partial S}{\partial z}\right) - S_{sinks} + S_{sources} \tag{11}$$

and the local salinity variation $\left(\frac{\partial S}{\partial t}\right)$ is determined by the advection and diffusion (small-scale motion) processes, and sources and sinks of salt (precipitation, evaporation, bottom springs and sinks). In this equation, the Fickian coefficients are, according to Osborne Reynolds in 1884, parameterized in terms of the small-scale velocity (u',v',w') and salinity fluctuations (S'):

$$K_x = -\frac{<u'S'>}{\frac{\partial S}{\partial x}}; K_x = -\frac{<v'S'>}{\frac{\partial S}{\partial y}}; K_x = -\frac{<w'S'>}{\frac{\partial S}{\partial z}}; \tag{12}$$

the symbol $< >$ indicates "time mean value" of the correlations of small-scale velocity components multiplied by the small-scale salinity variation. These values, multiplied by the density ($K_x = -\rho < u'S' >$; $K_y = -\rho < v'S' >$; $K_z = -\rho < w'S' >$), are the salt fluxes $[ML^{-2}T^{-1}]$ generated by turbulent diffusion.

The equations of motion are based on Newton's second law applied to the fluids, taking into account the forces and energy dissipation. The equation for a turbulent fluid was also presented in the classic paper of O. Reynolds. However, the estuarine water body has a particular geometry and has one open surface boundary; thus, special attention will be paid to the simplifications for their analytical and numerical solutions. For a hydrostatic fluid the equations of motion are:

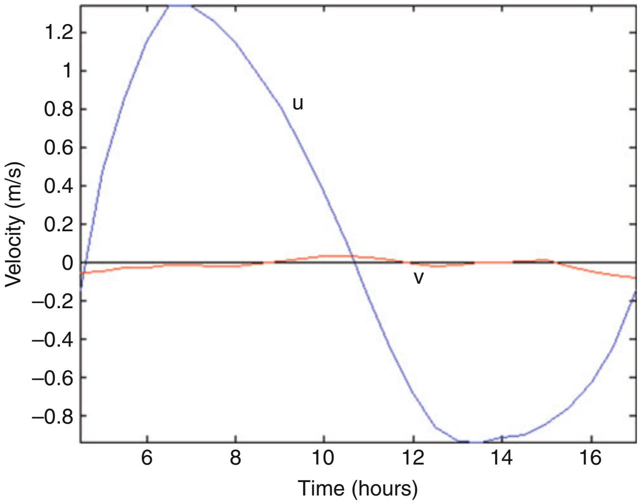
$$\frac{\partial u}{\partial t} + u\frac{\partial u}{\partial x} + v\frac{\partial u}{\partial y} w\frac{\partial u}{\partial z} - fv = -\frac{1}{\rho}\frac{\partial p}{\partial x} + \frac{\partial}{\partial x}\left(N_x\frac{\partial u}{\partial x}\right) + \frac{\partial}{\partial y}\left(N_y\frac{\partial u}{\partial y}\right) + \frac{\partial}{\partial z}\left(N_z\frac{\partial u}{\partial z}\right) \quad (13)$$

$$\frac{\partial v}{\partial t} + u\frac{\partial v}{\partial x} + v\frac{\partial v}{\partial y} w\frac{\partial v}{\partial z} + fu = -\frac{1}{\rho}\frac{\partial p}{\partial y} + \frac{\partial}{\partial x}\left(N_x\frac{\partial v}{\partial x}\right) + \frac{\partial}{\partial y}\left(N_y\frac{\partial v}{\partial y}\right) + \frac{\partial}{\partial z}\left(N_z\frac{\partial v}{\partial z}\right) \quad (14)$$

$$\frac{1}{\rho}\frac{\partial p}{\partial z} = -g \quad (15)$$

where f is the Coriolis parameter [$f = 2 \times \Omega\sin(\theta)$, Ω is the angular velocity of the earth and θ is the latitude], and g is the acceleration of gravity. In (11) and (14) N_x (K_x), N_y (K_y), and N_z (K_z) are the eddy kinematic viscosity (diffusion) coefficients. Since the estuarine water mass is assumed to be a system composed of pure water + salt, it will also be necessary to include in the hydrodynamic framework the equation of state of seawater, and the mass and salt conservation equations (4 and 11).

In analytical solutions, it will be assumed that the velocity does not change along its lateral axis (Oy), which is a good approximation because the secondary circulation intensity (v) is usually too low in comparison with the longitudinal (u) (Figure 3). Then, all terms of the preceding equations (11, 13, and 14) must be integrated along the Oy axis, and the mean value across its width (B) is calculated; thus, the equations are reduced to (Pritchard, 1958):



Estuarine Circulation, Figure 3 Time variations of the longitudinal (u) and the secondary (v) velocities components at the tropical Caravelas River Estuary (Bahia, Brazil) during spring tidal cycle of January, 2008.

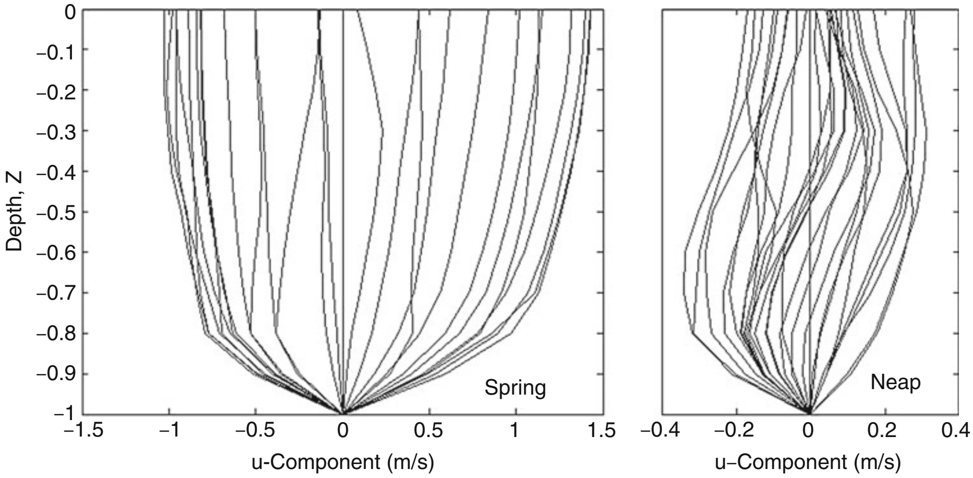
$$\frac{\partial u}{\partial t} + u\frac{\partial u}{\partial x} + w\frac{\partial u}{\partial z} = -\frac{1}{\rho}\frac{\partial p}{\partial x} + \frac{1}{B}\left[\frac{\partial}{\partial x}\left(BN_x\frac{\partial u}{\partial x}\right) + \frac{\partial}{\partial z}\left(BN_z\frac{\partial u}{\partial z}\right)\right] \quad (16)$$

$$\frac{\partial(uB)}{\partial x} + \frac{\partial(wB)}{\partial z} = 0 \quad (17)$$

$$\frac{\partial S}{\partial t} + u\frac{\partial S}{\partial x} + w\frac{\partial S}{\partial z} = \frac{1}{B}\left[\frac{\partial}{\partial x}\left(BK_x\frac{\partial S}{\partial x}\right) + \frac{\partial}{\partial z}\left(BK_z\frac{\partial S}{\partial z}\right)\right] - S_{sinks} + S_{sources} \quad (18)$$

In (16), (17), and (18), the quantities u, w, and S are mean values across the estuary width (B). This set of equations has the effect of decoupling the motion and mixing equations, i.e., the velocity components (u, w) obtained from the solutions of (16) and (17) are used in the (18) for the salinity profile solution. In the assumption that the width is constant (B = cte) and $N_z >> N_x$, and $K_z >> K_x$, these equations may be further simplified.

Equation (15) assumes that the hydrostatic balance and the expression of the horizontal gradient pressure force $\left(-\frac{1}{\rho}\frac{\partial p}{\partial x}\right)$ may be obtained in terms of its barotropic, baroclinic, and barometric components. On the assumption that the density (ρ) is known, the only unknown is the pressure (p), which may be easily obtained by vertical integration along the water column, from



Estuarine Circulation, Figure 4 Eulerian profiles of the u-velocity component at half-hour time intervals at spring (*left*) and neap tide (*right*) in the tropical Caravelas River Estuary (Bahia, Brazil). Note the *intratidal* and *subtidal* variabilities (From Andutta, 2011).

a depth z up to the sea surface slope η(x, y, t) (relative to a level surface):

$$p(x,y,z,t) = p_a(x,y,t) + g\int_z^{\eta} \rho dz \tag{19}$$

where p_a is the atmospheric pressure on the sea surface. Taking the differential of (19) using the Leibnitz differentiation rule it follows that:

$$\frac{\partial p}{\partial x} = \frac{\partial p_a}{\partial x} + g\left[\left(\rho_\eta \frac{\partial \eta}{\partial x}\right) + \left(\int_z^{\eta} \frac{\partial \rho}{\partial x} dz\right)\right] \tag{20}$$

where ρ_η is the density on the surface. From this result, with $\frac{\rho_\eta}{\rho} \approx 1$, the expression for the longitudinal component of the gradient pressure force (per mass unit) has three components:

$$-\frac{1}{\rho}\frac{\partial p}{\partial x} = -\frac{1}{\rho}\frac{\partial p_a}{\partial x} - g\frac{\partial \eta}{\partial x} - \frac{g}{\rho}\int_z^{\eta} \frac{\partial \rho}{\partial x} dz \tag{21}$$

namely, barometric (a), barotropic (b), and baroclinic (c), respectively:

(a) The barometric component is related to transient weather systems (typically 3–10 days) associated with low-pressure centers and has subtidal variability. Under steady state, the sea surface acts as an inverted barometer; for $\Delta p = \pm 1.0$ mbar the sea surface decreases/increases by 1.0 cm. However, if a storm surge was to reach an estuary, it may cause severe dangerous floods, especially during spring tide.

(b) Is independent of depth and varies according to the sea surface slope oscillation. In normal tidal conditions, its highest and smallest values occur during the spring and neap tide, respectively. Its order of magnitude varies approximately in the interval -10^{-3} – $+10^{-3}$ (ms^{-2}). Thus, it is considered to have inter or subtidal variability.

(c) This component is zero on the surface ($z = \eta$) and increases with the depth, up to a magnitude order of -10^{-4} ms^{-2}. Due to the inter and subtidal variability of the density field, its numerical value is not an easy quantity to be determined.

During the flood tides, barotropic (b) and baroclinic (c) forces act up-estuary, but during the ebb they act in the opposite direction.

In the variability analysis of the half-hourly Eulerian profiles at the spring tide (Figure 4-left), the higher barotropic tidal forcing, generating bidirectional motions up to 1.5 and −1.0 ms^{-1} (ebb and flood, respectively) preclude the baroclinic forcing, but the opposite occurs during the neap tide (Figure 4-right). The action of the less intense baroclinic pressure force is clearly seen in generating bidirectional motions and in the speed increase (in intensity) at mid-depths during the flood ($u < 0$).

In analytical solutions, the expression of the baroclinic component may be simplified on the assumption that it is independent of the depth $\left[\frac{\partial}{\partial z}\left(\frac{\partial \rho}{\partial x}\right) = 0\right]$, by using a depth time-mean estimated value $\left(\frac{\partial \rho}{\partial x} = \rho_x\right)$. Then, for a simple geometry (B = cte), kinematic eddy viscosity coefficient N_z independent of depth and $N_z >> N_x$, the simplified steady-state equation of motion is:

$$-g\frac{\partial \eta}{\partial x} - \frac{g}{\rho}\rho_x(z+h) + N_z\frac{\partial^2 u}{\partial z^2} = 0, \tag{23}$$

which states a balance between the barotropic and baroclinic components of the pressure gradient force and the vertical velocity shear associated with the estuarine circulation; the influence of tides within this formulation enters only in the value of N_z (Geyer, 2010).

Analytical and numerical solutions

Mathematical models can be either analytical or numerical Ji (2008). An analytical model has an exact mathematical solution to the differential equations describing processes in estuaries Blumberg (1975). They may be applied to relatively restrictive conditions, usually for one or two dimensions, constant parameters and steady-state conditions. In spite of the severe assumptions that must be invoked, analytical models are often used to (Ji, 2008):

1. Check the accuracy of numerical models (e.g., Blumberg, 1975).
2. Provide first-order estimates of relatively simple systems.
3. Give insights into hydrodynamic and water quality processes in estuaries.

A numerical model is a discretized version of a set of mathematical equations, as presented in this chapter (continuity, equation of motion, salt conservation), which describes processes in the estuary, and can be implemented as a computer program. By entering the input data and mode parameters into the computer model, numerical and graphical simulations of an estuary, in response to a set of forcing conditions and boundary conditions, may be obtained.

However, analytical and numerical model solutions must be calibrated or validated based on observational data. This may be done numerically with nondimensional parameters such as the Relative Mean Absolute Error (Walstra et al., 2001) and the Skill parameter (Wilmott, 1981), further applied by Warner et al. (2005). The vertical mean Skill parameter was adapted by the validation of vertical velocity and salinity profiles (Andutta et al., 2006). The Skill parameter is calculated taking into account the model solutions and the observational data and varies from 1 to 0 (zero) indicating the best fit and a complete disagreement between observation and the theoretical results, respectively.

Analytical models

The first steady-state analytical model for determining time mean longitudinal velocities in a coastal plain estuary was developed by Pritchard and Kent (1956) using the lateral and longitudinal components of the equation of motion, the tidal velocity amplitude, and the relationship between the vertical and lateral eddy stress. The method was applied to stations in the James River Estuary, studied in detail during several tidal cycles in the summer (June and July, 1950). The theoretical velocity profiles agreed well with the observational data, showing typical velocity profiles of a partially mixed estuary – seaward and up-estuary motion on the upper and lower layers, respectively, and no motion at mid-depths. The Pritchard and Kent paper is a pioneering article showing the importance of the comparison of theoretical versus experimental data.

Hansen and Rattray (1965) developed a steady-state analytical model for the circulation and mixing of partially mixed estuaries. The laterally averaged equations of motion and the mass and salt conservation equations (16, 17, and 18) were used with the simplifications: $\left(\frac{\partial u}{\partial \mathrm{x}} << \frac{\partial \mathrm{u}}{\partial \mathrm{z}}\right)$, $\left[\frac{\partial}{\partial \mathrm{z}}\left(\frac{\partial \mathrm{S}}{\partial \mathrm{x}}\right) = 0\right]$ and a linear equation of state of seawater $\rho = \rho(\mathrm{S})$ for the hydrodynamic equations closure. Using similarity solution techniques, the model considers the balance of the barotropic and baroclinic modes, with wind stress forcing on the surface ($\mathrm{u} = 0$) and no-sleep condition at the bottom $[\mathrm{u}(-\mathrm{h}) = 0]$. The central regime solutions for $\mathrm{u} = \mathrm{u}(\mathrm{z})$ and $\mathrm{S} = \mathrm{S}(\mathrm{x}, \mathrm{z})$ depend on subjective numerical values such as the longitudinal density gradient and the mean salinity at the mouth, as well as the eddy viscosity and diffusion coefficients. The results of this model used for the Piaçaguera channel (upper reaches of the Santos Channel – São Paulo – Brazil), validated with the mean vertical Skill parameter, are presented in Figure 5, according to Miranda et al. (2012).

A simple and direct solution to (23), conducive to the same result of Hansen and Rattray (Figure 5), was given by Officer (1976). On the assumption that the longitudinal density gradient ρ_x is a known quantity, the equation has two unknowns: $\left(\frac{\partial \eta}{\partial \mathrm{x}}\right) = \eta_x$ and $\mathrm{u} = \mathrm{u}(\mathrm{z})$. Thus, a second equation is necessary to complete the equation system. In this solution, the equation of continuity integrated in the estuary volume, called the integral boundary condition, is used:

$$\frac{1}{\mathrm{h}} \int_{-\mathrm{h}}^{0} \mathrm{u}(\mathrm{z})\mathrm{dz} = \frac{\mathrm{Q_f}}{\mathrm{A}} = \mathrm{u_f} \tag{24}$$

where Q_f is the river discharge, and A, h, u_f are the cross-section area, the depth, and the fresh water velocity, respectively.

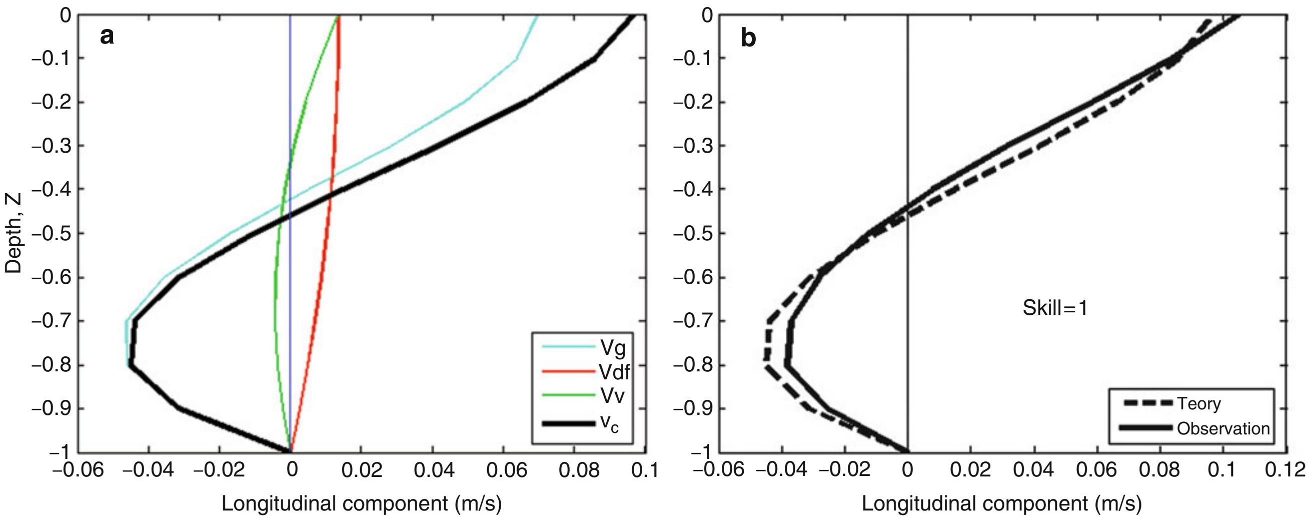
To achieve the solution, the upper and the lower boundary conditions are the same as in the Hansen and Rattray analytical model: $\rho N_z \frac{\partial \mathrm{u}}{\partial \mathrm{z}}\big|_{\mathrm{Z}=\eta} = -\tau_{\mathrm{W}}$ and $\mathrm{u}|_{\mathrm{z}\,=\,-\mathrm{h}} = 0$

For (23), which is a second order ordinary differential equation, its general solution will be dependent on two integration constants C_1 and C_2,

$$\mathrm{u}(\mathrm{z}) = \frac{1}{2}\frac{\mathrm{g}}{\rho N_z}\eta_x \mathrm{z}^2 + \frac{\mathrm{g}}{\rho N_z}\rho_x\left(\frac{\mathrm{h}}{2}\mathrm{z}^2 + \frac{1}{6}\mathrm{z}^3\right) + C_1\mathrm{z} + C_2 \tag{25}$$

which are determined according to the upper and lower boundary conditions:

$$C_1 = \frac{\tau_{\mathrm{W}}}{\rho N_z}, \text{ and } C_2 = -\frac{\tau_{\mathrm{W}}\mathrm{h}}{\rho N_z} - \frac{1}{2}\frac{\mathrm{g}\eta_x}{N_z}\mathrm{h}^2 - \frac{\mathrm{g}}{3\rho N_z}\rho_x \mathrm{h}^3 \tag{26}$$



Estuarine Circulation, Figure 5 (**a**) Theoretical results for each component mode: Vg = baroclinic forcing; Vdf = river discharge; Vv = wind stress and Vc = composite profile, in the Piaçaguera channel in the upper reaches of the Santos channel (São Paulo – Brazil, using the Hansen and Rattray (1965) analytical model. (**b**) Experimental versus observational of the u-velocity profile. Skill is the mean vertical parameter to validate the theoretical simulation (From Miranda et al., 2012).

if these values are known, the general solution is:

$$u(z) = \frac{g}{\rho N_z}\rho_x\left(\frac{z^3}{6}+\frac{h}{2}z^2-\frac{h^3}{3}\right)+\frac{1}{2} \times \frac{g}{N_z}\eta_x\left(z^2-h^2\right)+\frac{\tau_W}{\rho N_z}(z-h) \qquad (27)$$

or in nondimensional depth $Z=\frac{z}{|h|}$,

$$u(Z) = \frac{g}{6N_z\rho}\rho_x h^3\left(Z^3+3Z^2-2\right)+\frac{1}{2} \times \frac{g}{N_z}\eta_x h^2\left(Z^2-1\right)+\frac{\tau_W h}{\rho N_z}(Z+1) \qquad (28)$$

This solution is still dependent on the second unknown (η_x) and we must apply the integral boundary condition, written in terms of the nondimensional depth Z:

$$\int_{-1}^{0} u(Z)dZ = \frac{Q_f}{A} = u_f, \qquad (29)$$

and the result for the surface slope η_x is:

$$\eta_x = -\frac{3N_z}{gh^2}u_f - \frac{15}{24}\frac{\rho_x}{\rho}h + 3\frac{\tau_W}{\rho g h} \qquad (30)$$

and depends on three quantities: (1) the fresh water velocity, (2) the baroclinic component and (3) the wind stress. An order of magnitude analysis indicates that the baroclinic term, associated with the longitudinal density gradient, is the dominant equivalent to the M. Margules rule for the slope of an interface in the atmosphere, which was adapted to oceanographic use by A. Defant in 1929 (quoted in von Arx, 1962, 383).

Combining (28) and (30), the final solution for u = u (Z) is Officer (1976):

$$u(Z) = \frac{gh^3}{48N_z\rho}\rho_x\left(8Z^3+9Z^2-1\right)+\frac{3}{2}u_f\left(Z^2-1\right)+\frac{1}{4} \times \frac{h}{\rho N_z}\tau_W\left(-3Z^2-4Z-1\right) \qquad (31)$$

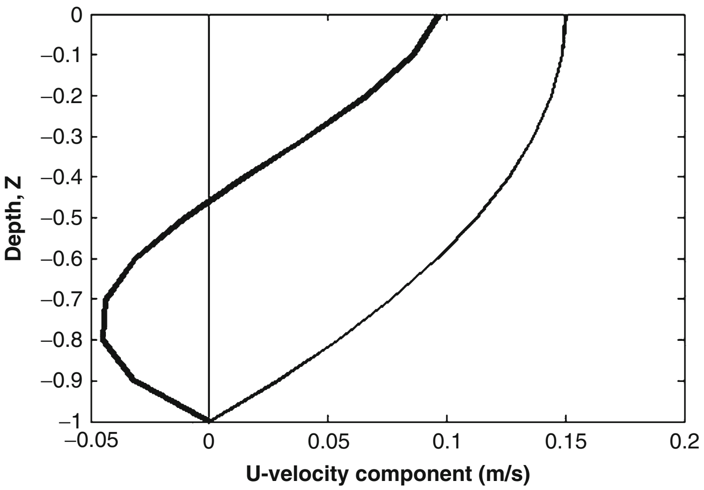
and its graphical results agree with those of Hansen and Rattray (1965) shown in (Figure 5b).

From (31), it is possible to simulate the seaward and unidirectional circulation of a well-mixed estuary ($u > 0$) by changing parameters values as increasing the water column height (h), decreasing the longitudinal density gradient (ρ_x) and the kinematic viscosity coefficient (N_z) (Figure 6).

Further classical and up-to-date analytical solutions of salt wedge, well-mixed, and partially mixed estuaries may be found in Prandle (2009) and Miranda et al. (2012).

Secondary circulation

The secondary estuarine circulation is normal to the along-channel currents and is an integrated component of the estuarine circulation. Its dynamics have been presented in several articles since the pioneering works by Okubo (1973). Taking into account experimental results Dyer (1977), presented diagrammatic representations of the



Estuarine Circulation, Figure 6 Theoretical velocity profiles for a partially mixed estuary (tick line – the same as in Fig. 5b) and the unidirectional motion of a well-mixed estuary with ebbing currents ($u > 0$ – thin line).

mean along-channel currents and the associated secondary circulation for different estuarine types: salt-wedge (types A or 4), partially mixed (types B or 2), and well mixed (types C or 1). The dynamic balance of the secondary flows was established by taking into account the interaction of the following factors: bottom topography and channel geometry, lateral stratification of density due to vertical mixing, and gradient pressure forces, Coriolis acceleration and centrifugal acceleration (Nunes and Simpson, 1985).

In the bi-dimensional analytical simulation of the secondary circulation, according to Nunes and Sympson (op. cit.), the simplifying hypothesis are applied: (1) stationary conditions; (2) lateral density (salinity) stratification $\frac{\partial\rho}{\partial y} = g(y)$, and; (3) straight channel. The solution of the secondary circulation $v = v(y, Z)$ and its associated vertical component $w = (y, Z)$ obtained with this model is exemplified by Miranda et al. (2012).

The importance of the secondary flows and mixing across a channel as interrelated processes, were recently discussed in detail by Chant (2010).

Summary

Estuaries were formed in a narrow coastal boundary zone between the sea and land, during the last interglacial period. This contribution is intended to present some aspects of estuarine circulation related to: (1) transitional environments along the coastline that are ultimately dependent on estuarine dynamics; and (2) investigations of the processes that have focused on estuarine preservation, water quality, morphology, biodiversity, and fisheries which are strongly dependent on the dynamic behavior of estuaries.

Some focus is given to the pioneer research published in the last half of the twentieth century. Additionally, experimental and theoretical developments have recently been published focusing on the following issues: estuary definition, classification, mixing processes, variability, and circulation.

Transitional environments along the coastline are ultimately dependent on the dynamics of estuarine systems; however, they have been strongly affected by human activities. Some of them have been protected by reserve status since the eighteenth century. Human impacts and their effects on estuaries and coasts are important factors that need to be assessed.

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ESTUARINE CONNECTIVITY

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Definition

Connectivity is defined as the state of being connected or interconnected (Oxford Dictionary). From an ecological perspective, it is better described as an empowering mechanism that facilitates the movement of materials or effects between spatiotemporal units and enables events in one spatiotemporal unit to influence events in another unit.

Estuarine connectivity is a facilitator of the movement of materials or effects in an estuarine context. It occurs at multiple scales: within the estuary, between the estuary and other contiguous marine and terrestrial systems, and between habitats within the estuary.

Introduction

Estuaries are dynamic ecosystems that interact with both the ocean downstream and freshwater from rivers upstream. The pattern of dilution varies between different estuaries and depends on the volume of freshwater, the tidal range, and the extent of evaporation of the water in the estuary (McLusky and Elliott, 2004). It is connectivity that allows the interactions to occur. Connectivity occurs at all scales: between estuaries and other ecosystems but also among different habitats within estuaries, for example, among seagrass, mangrove, and saltmarsh systems.

Connectivity is a facilitator of the complex interaction between biogeographic and physical variables (tides, salinity, DO, pH), ecological processes (larval supply, competition, predation, nutrient dynamics), and evolutionary processes (gene flow). Table 1 shows some examples.

Estuarine Connectivity, Table 1 Examples of some types of estuarine connectivity

Connectivity type	Example	References
Genetic	Gene flow between populations of an estuarine fish	Bradbury et al. (2008)
Life history	Fish use of estuaries as nursery grounds	Sheaves et al. (2007a), Davis et al. (2012)
Temporary access to habitat	Fish accessing saltmarsh on flooding tide and leaving on the ebb	Minello et al. (2003), Meynecke et al. (2008)
Nutrient/sediment flows	Movement of nutrients and sediments from freshwater to the estuary	Wolanski (1995)
Nutrient subsidy	Movement of nutrients among habitats (e.g., from seagrass, thereby sustaining adjacent habitats)	Connolly et al. (2005)

Connectivity not only facilitates processes but also determines if they can occur. It is often asymmetric. That is, the linkages between places and between organisms are not necessarily equal in both directions. This was noted in Beger et al. (2010b) in the context of conservation planning and is applicable to estuaries. Examples would include nutrient and sediment flows in an estuary from upstream that have no upstream return.

The connectivity medium provides the potential for connection to occur. Whether or not actual connectivity occurs depends on other factors. For instance, if oxygen limits connectivity, it may prevent access by fish; if there are alternative food sources, some connections are not needed or used at a particular time.

Connectivity operates in space and in time and both may interact. For example, seasonally disconnected estuaries (intermittently closed and open lakes and lagoons [ICOLLs]) may connect to marine influences but not all the time, particularly where the tidal range is small, as conditions can be highly variable (Gale et al., 2006). Where tidal ranges are larger, Abrantes and Sheaves (2010) showed that hydrologic connectivity was key to maintaining, for example, carbon sources over a range of estuarine types.

Connectivity not only affects mobile organisms that are able to respond to short-term changes in connectivity but also affects vegetation that may respond to longer-term changes in connectivity. As an example, mangroves in eastern Australian estuaries have been encroaching into saltmarsh over several decades, with rainfall as one of the main drivers (Eslami-Andargoli et al., 2009) and wetland topography providing the opportunity for connection.

Classes of connectivity

Classes of connectivity include both narrow and broad interfaces and also constrained and diffuse connections (Beger et al., 2010a; Beger et al., 2010b). The two categories especially relevant to estuaries are the broad interface and diffuse connection types. The broad interface category in estuaries has a mixing of fresh- and saline water and so is associated with organisms that require brackish conditions, that is, their habitats experience the effects of both sea- and freshwater. The diffuse connection category in estuaries is variable with both fresh and saline components that are spatially disjunct but which nevertheless facilitate connectivity for organisms that spend different parts of the life cycle in contrasting habitats – needing saline or freshwater at specific stages.

Threatening processes: natural and anthropogenic

Natural processes that can threaten estuarine connectivity include climate changes such as increasing extreme events that can, over a short time frame, destroy fringing mangroves in tropical estuaries and tall grasslands in temperate ones. Sea-level change is also likely to affect estuarine connectivity, changing the patterns of interaction between fresh- and saline waters and thereby having critical impacts on aquatic biota including vegetation.

Human activities can interrupt connectivity. For example, constructing barriers, dams, and hydroelectric schemes impacts organisms that depend on connections between saline and freshwater habitats. Examples include the effects of barriers on marine-spawned fish in South Africa (Wasserman et al., 2011) and on fish biota throughout a temperate system in Australia (Rolls, 2011), the impact of impoundments on estuaries in Australia (Sheaves et al., 2007b), and the impacts of hydropower developments in Vietnam (Sheaves et al., 2008). These are all point sources of interruption. There are also the more diffuse effects of development that can change, for example, surface hydrology, sediment and nutrient supply and which can affect both the broad interface extent and the diffuse pattern of salinity types, with ongoing impacts on dependent organisms.

Summary

Estuarine connectivity facilitates biogeographic, ecological, and evolutionary processes within and among estuarine systems and between estuarine systems and their upstream fluvial and terrestrial ecosystems and their downstream marine ones.

Estuarine connectivity is crucial to sustaining ecosystems but vulnerable to threatening processes of natural or anthropogenic origin at a variety of scales.

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Cross-references

Climate Change
Coastal Lagoons
Coastal Wetlands
Dissolved Oxygen
Estuarine Circulation
Food Web/Trophic Dynamics
Forested Wetland Habitat
Mangroves
Saltmarshes
Sea-Level Change and Coastal Wetlands
Sediment Transport
Tidal Ranges
Wetlands

ESTUARINE DELTAIC WETLANDS

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Definition

Wetlands developed on a delta within an estuary.

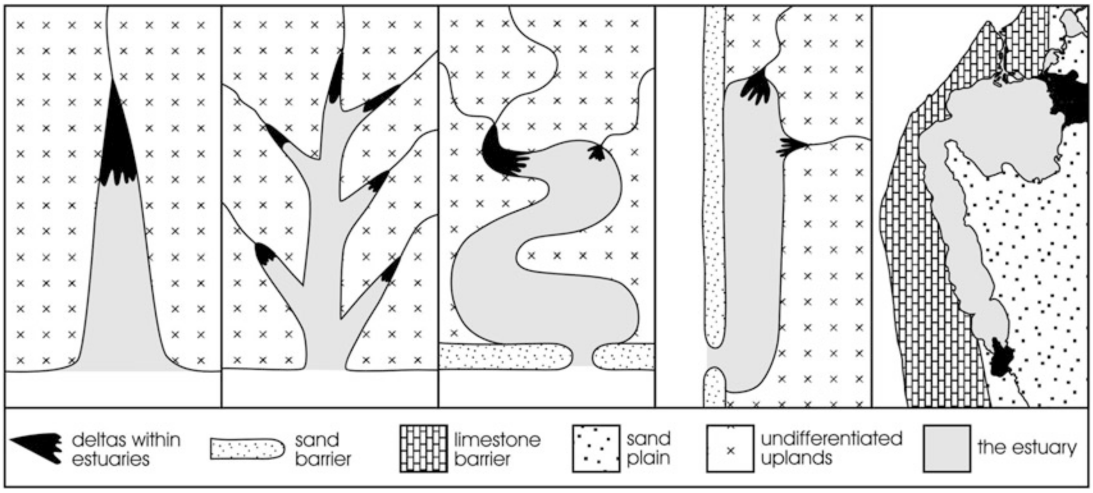
An estuary and the formation of a delta

An estuary is a river-to-marine transitional environment where marine salinity is measurably diluted by (riverine) freshwater in a valley tract, an inlet, a coastal lagoon, or an embayment, producing a salinity gradient from the river to the sea, from freshwater to marine. This is a *positive estuary.* In some climates, where river contribution is seasonal or is markedly less than the evaporation from the coastal water body, the inner parts of the estuary are subject to evaporation, and the head of the estuary may become seasonally hypersaline or at least attain salinity higher than seawater. This is a *negative* or *inverse estuary.* In either case, freshwater influx usually is via a river.

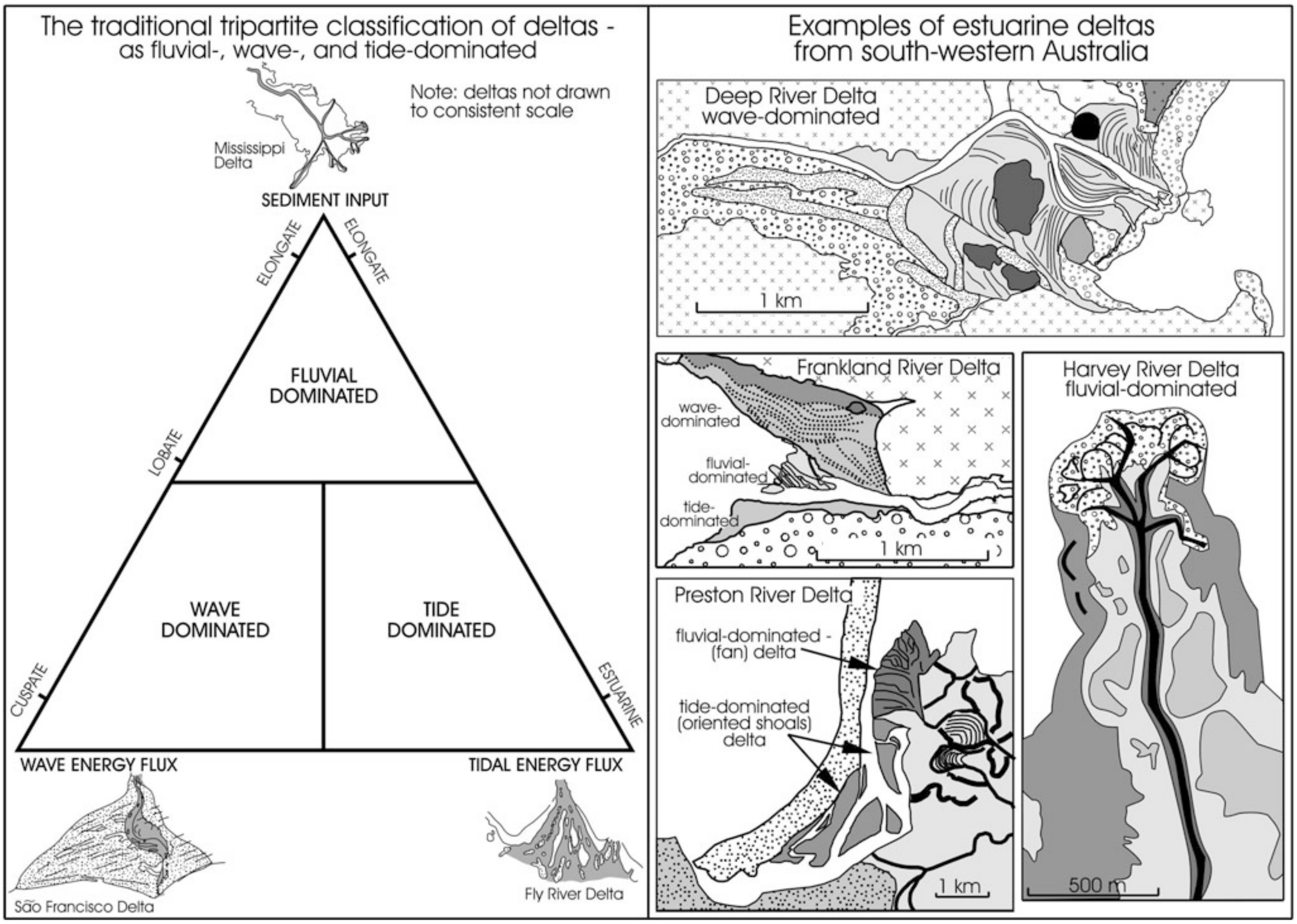
In the most common context where the freshwater delivery system to the estuary is riverine, the river also delivers fluvial sediment as traction load of sand and gravel or in suspension as silt and clay. Such sedimentary material commonly is deposited at the mouth of the river where it enters the more open water system of the estuary and forms a delta (Figure 1). Elliott (1986) defines a delta as "a discrete shoreline protuberance formed where a river enters an ocean, a semi-enclosed sea, a lake or lagoon, and supplies sediment more rapidly than it can be redistributed by basinal processes." This definition incorporates deltas formed in estuarine bays, estuarine embayments, and estuarine lagoons.

Types of estuarine deltas

Deltas are common landforms within estuaries, generally occurring at the head of an estuary which is the terminal end of the contributing fluvial valley system. With complex estuaries, or estuaries with multiple tributaries, there may be multiple deltas along the estuary margins as well as at its head. The deltas, depending on the volume of freshwater influx and amount of fluvial sediment delivered, may be of various sizes. Rivers with a voluminous sedimentary budget will develop large deltas. Depending on their location within the estuary and the hydrodynamics of the estuarine water body where the delta is located, the delta will be shaped by forces of river flow, waves, and tides into a fluvial-dominated form, wave-dominated form, or tide-dominated form (Figure 2; Wright and Coleman, 1973; Galloway, 1975; Reineck and Singh, 1980). For instance, the delta, facing a large fetch across the estuarine water body, where there are strong cross-estuary winds forming prevailing wind



Estuarine Deltaic Wetlands, Figure 1 Idealized diagram showing a range of estuary types, from an incised single valley to rias, a flooded valley on a coastal plain, a barred estuarine coastal lagoon, and a compound estuary, and the occurrence of estuarine deltas (*black*) therein.



Estuarine Deltaic Wetlands, Figure 2 Ternary diagram summarizing the types of deltas formed under conditions of waves, tides, and fluvial regimes (Modified after Reineck and Singh, 1980) and examples of deltas as wave-dominated, tide-dominated, and fluvial-dominated forms within some estuaries of southwestern Australia (Deep River Delta and Frankland River Delta of Walpole-Nornalup Inlet Estuary, Harvey River Delta of Peel-Harvey Estuary, and Preston River Delta of Leschenault Inlet Estuary).

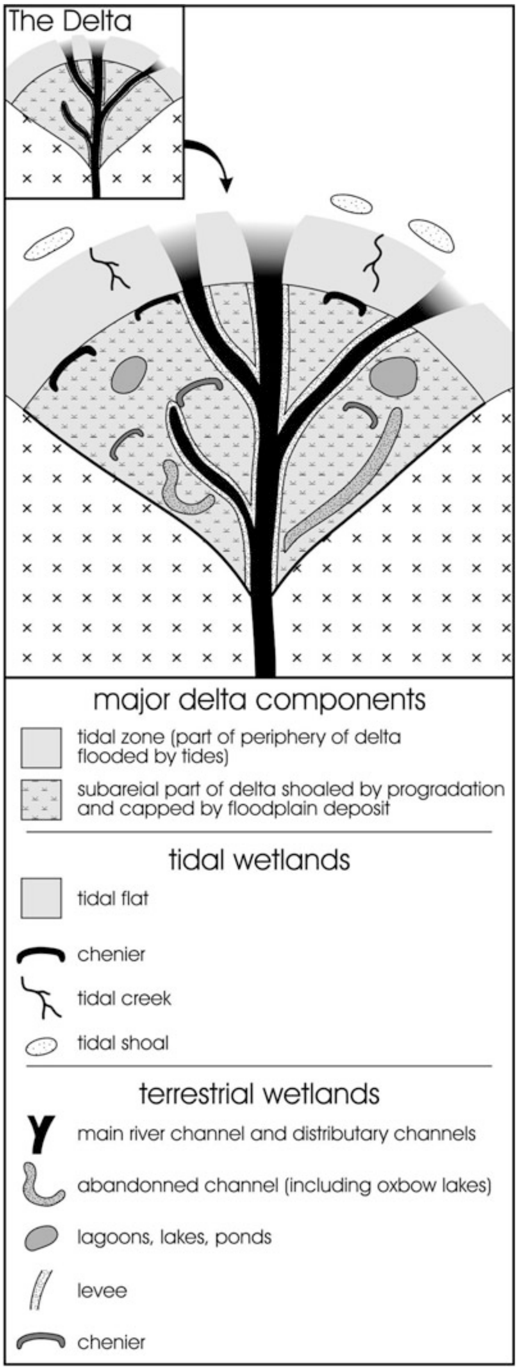
waves, will have a wave-dominated form. Alternatively, where tide and wave energy are relatively low, the delta will have a fluvially dominated form.

Landforms of deltas that can form wetlands

The landforms that comprise deltas form the foundation to wetlands and the habitats for wetland biota. Landforms vary across the different types of deltas and across a single delta type. In fluvially dominated deltas, the landforms comprise tidal flats, beaches, tidal creeks, beaches, floodplains, local cheniers, distributary channels, levee banks, water-filled abandoned channels, and sediment-filled abandoned channels. In wave-dominated deltas, the landforms comprise bars, beaches, sandy tidal flats, beach ridges, swales, narrow to circular lagoons leeward of bars and beach ridges, tidal flats, distributary channels, rarer tidal creeks, levee banks, water-filled abandoned channels, sediment-filled abandoned channels, and floodplains. In tide-dominated deltas, the landforms comprise muddy and sandy tidal flats, tidal shoals, tidal creeks, floodplains, local cheniers, distributary channels, levee banks, water-filled abandoned channels, and sediment-filled abandoned channels. In broad terms, the delta can be separated into a subaerial portion that has aggraded above tidal levels and largely encompasses fluvial landforms and processes, and into a tidal portion that encompasses marine/estuarine landforms and processes.

What is a wetland on a deltaic plain?

In a delta, a wetland essentially is either the terrestrial or the tidally inundated part of the delta that is wet by natural processes such that there is a permanent physical, chemical, or biological imprint on the wetland sediment or soil reflecting the hydric or hydrochemical condition, or a biological response in terms of growth of hydrophilic vegetation or accumulation of shell (Figure 3). For terrestrial wetlands on the subaerial parts of deltas (i.e., those



Estuarine Deltaic Wetlands, Figure 3 Idealized illustration showing the separation of terrestrial (subaerial) wetlands on the delta and wetlands on the tidally inundated part of the delta. Some of the more common wetland types on the subaerial (terrestrial) part of the delta are diagrammatically shown.

Estuarine Deltaic Wetlands, Table 1 Classifying terrestrial/subaerial deltaic wetlands according to their landform and hydrology (classification categories applicable to delta systems, after Semeniuk and Semeniuk 2011)

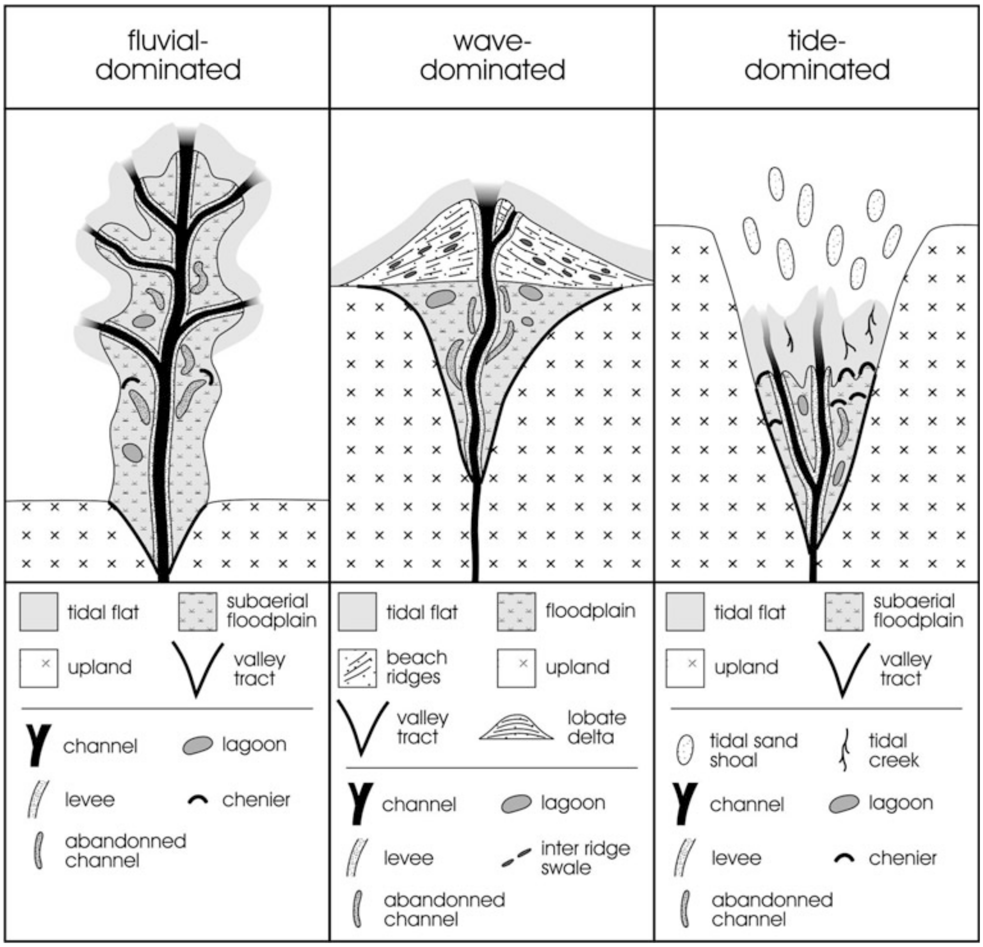
	Landform		
Hydrology	Flat	Basin	Channel
Permanently inundated	[a]	Lake	River
Seasonally inundated	Floodplain	Sumpland	Creek
Seasonally waterlogged	Palusplain	Dampland	Trough
Intermittently inundated	Barlkarra	Pirapi	Wadi

[a] Not applicable

parts of the delta that are distant enough from the estuarine aquatic environment such that they are not wet by the tides and waves of the estuarine aquatic environment), the wetting process(es) may be solely one or a combination of the following: (1) rain (where the rainwater is ponded and perched or waterlogs the surface materials to form the wetland); (2) water table rise and fall in response to tide fluctuation, or to seasonal rain, or to seasonal regional throughflow; (3) the effect of a shallow water table resulting in wetting of surface sediments/soils by capillary rise; (4) over-bank flow from the distributary channels; and (5) run-in where surface flows collect into a basin or a water-retaining landform such as a floodplain. For wetlands on the tidally inundated part of the delta that form the "tidal coast" fronting the estuarine aquatic environment and are subject to wetting by coastal processes of tides and waves, the wetting process (es) may be one of the following: (1) daily wetting of tidal flats by tides, (2) the daily flooding and draining of tidal creeks by the tide, (3) wave swash and run-up, and (4) storm surges and atmospheric depression-induced high-water stands causing flooding of lowlands by incursion of estuarine water or marine water. As such, all terrain built by fluvial processes to levels above the high tide that are wet by fluvial and meteoric processes are terrestrial wetlands, and all surfaces that are daily flooded and exposed by tides or wet by wave action and storm surges are the tidally inundated wetlands on the delta. It should be noted that all deltaic surfaces that are permanently subtidal are not considered to be wetlands sensu Semeniuk and Semeniuk (1995, 2011).

Depending on the wetting agent – rain, river water, estuarine water, or marine water – and on the frequency of the wetting process, the groundwater and/or surface water of the wetland may be freshwater, brackish, or saline, and it may be stasohaline (remaining relatively constant throughout the year) or poikilohaline (e.g., alternating between salinity fields, such as between freshwater and saline water).

Individual wetlands can be classified according to their geomorphic and hydrologic characteristics, following Semeniuk and Semeniuk (1995, 2011) for nonmarine wetlands and Semeniuk (1986) for coastal tidally influenced wetlands. The wetland classification of Semeniuk and Semeniuk (1995, 2011), termed the "geomorphic-hydrologic classification," recognizes the various types of landforms that comprise wetlands and the hydrologic



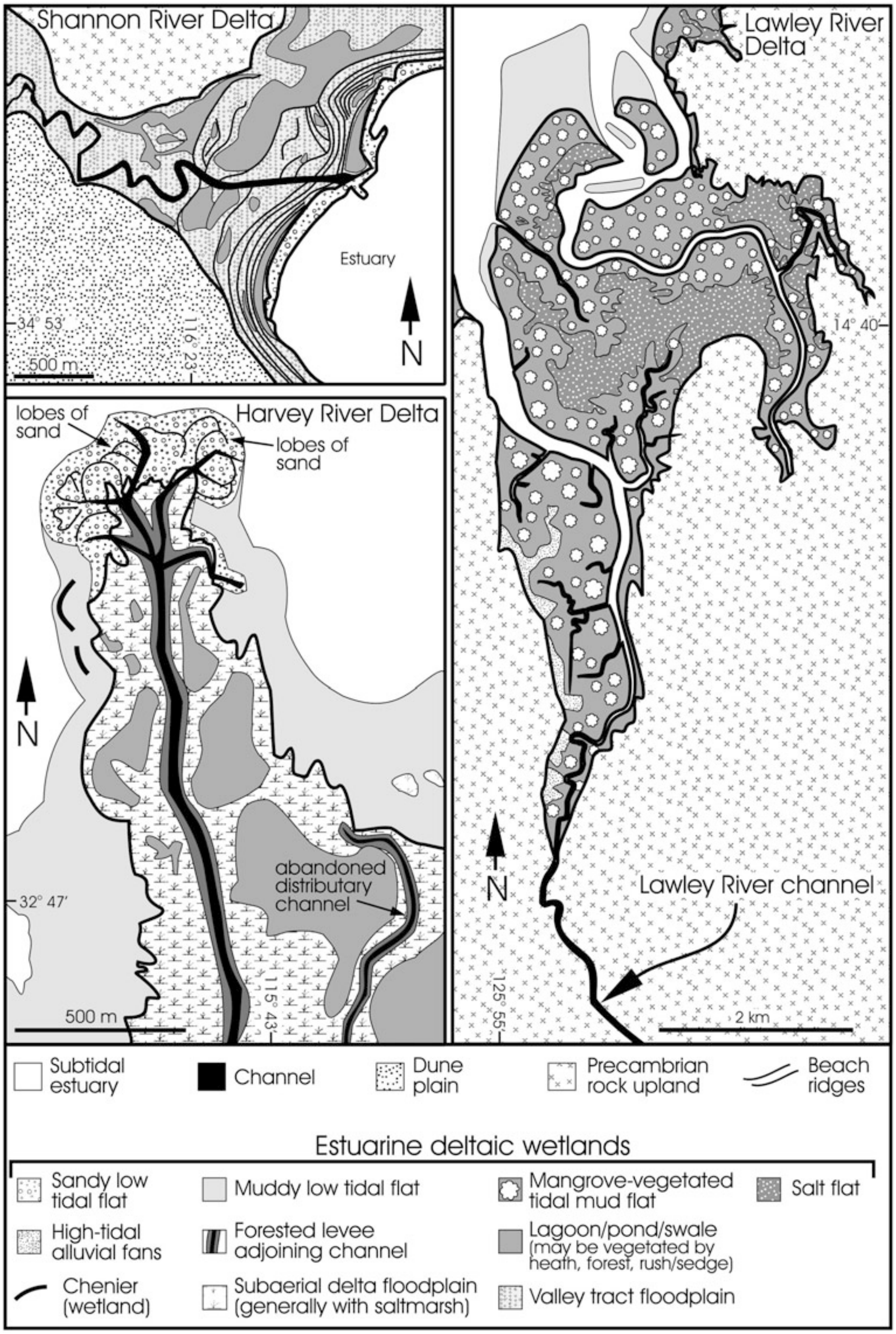
Estuarine Deltaic Wetlands, Figure 4 Idealized diagram showing the distribution and range of wetland types that can occur specific to a fluvial-dominated delta, a wave-dominated delta, and a tide-dominated delta.

regime that maintains (or wets) them in terrestrial settings. In deltas, for instance, the landforms comprising wetlands are mainly basins, flats, and channels, and the hydrological regimes that maintain them are permanent inundation, seasonal inundation, seasonal waterlogging, intermittent inundation, and daily to fortnightly tidal inundation. For the terrestrial component of deltas, combining landform type with hydrologic regime results in categories of wetlands, which are then assigned nomenclature (or terms) as shown in Table 1. The wetland categories are further classified with descriptors as to size, shape, and water salinity (Semeniuk and Semeniuk, 1995, 2011). For the tidally inundated part of the deltas, the wetlands are identified as geomorphic units, viz., tidal flats, beaches, tidal creeks, and spits and cheniers. Cheniers, depending on the level to which that have aggraded, may be inundated by tides or may have been constructed beyond the level of tide influence and may be inundated only by the highest river floods. As such, they can belong either to the subaerial or to the tidal realm of a delta.

The main categories of wetlands (and the equivalent formal nomenclature term in brackets) that are common within a delta using Semeniuk and Semeniuk (1995, 2011) and Semeniuk (1986) are permanently inundated basins such as water-filled abandoned channels and water-filled lagoons (lakes), seasonally inundated basins such as seasonally water-filled abandoned channels and water-filled lagoons (sumplands), seasonally waterlogged basins (damplands), seasonally inundated flats such as floodplains adjoining the channels (floodplains), seasonally waterlogged floodplains (palusplain), permanent channels (rivers), seasonally flowing channels (creeks), tidally flooded flats (tidal flats), and tidally flooded channels (tidal creeks).

The delta as a wetland complex

The processes of delta construction result in terrain which is essentially a nearly flat, low-lying surface close to a water table. Deltas are sedimentary deposits that either have shoaled to (at least) the level of the high tide or have



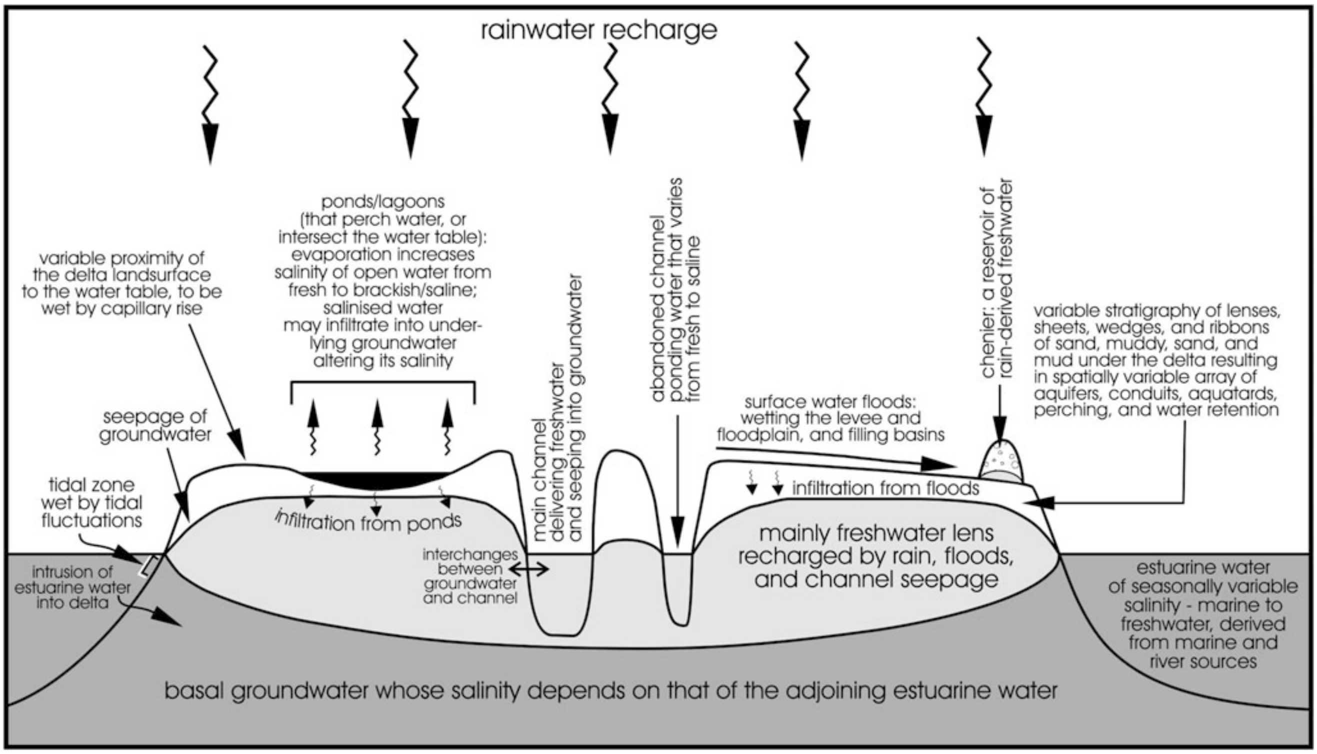
Estuarine Deltaic Wetlands, Figure 5 Distribution of wetland types in a wave-dominated delta, a fluvial-dominated delta, and a tide-dominated delta in the Broke Inlet Estuary (Shannon River Delta), the Peel-Harvey Estuary (Harvey River Delta), and the Lawley River Estuary (Lawley River Delta), respectively, in Australia.

Estuarine Deltaic Wetlands, Table 2 The range of wetlands commonly occurring in estuarine deltas

Landform type	Wetland type	Description and origin
Terrestrial or subaerial part of the delta (i.e., not tidally flooded)		
Inter-swale lagoons	Lake, if permanently inundated; sumpland, if seasonally inundated	Water-filled linear basin (swale) formed between beach ridges; the swales or lowlands between beach ridges that are inundated by floods, or have floors that intersect the water table, or are wet by proximity to a water table
Swale basins leeward of sand bars and beach ridges, but not inundated by tides	Lake if permanently inundated, sumpland or dampland if seasonally inundated or seasonally waterlogged	Basin formed leeward of a sand bar or beach ridge and whose floor is sandy, peaty sand, peat, terrigenous mud, or carbonate mud; the basins are inundated by floods, or have floors that intersect the water table, or are wet by proximity to a water table
Sediment-filled abandoned channels	Usually a sumpland, sometimes a lake	Water-filled linear basin (the abandoned distributary channel) filled with sand, mud, or peat; water in the channel derives from river floods or rainfall
Supratidal flats	Supratidal flat	Low gradient sloping flat underlain by sand or mud, located above the level of the highest tide, formed by progradation, and inundated usually by river floods
Floodplains (river-flood levels)	Floodplains	Flat surfaces underlain by sand, muddy sand, or terrigenous mud formed by accretion of floodwater sediment
Flats, basins, and swales proximal to a water table	Palusplains and damplands	Flat surfaces or basins underlain by sand, muddy sand, or terrigenous mud; flooding by river overflow or by rainfall
Levee bank	Low relief emergent landform	Convex surface underlain by fine sand or silt, flooded annually or intermittently
Spit or chenier	Low relief emergent landform	Convex surface underlain by sand or shell and sand whose crest, if above the highest tides, is inundated by high river floods
Tidally inundated part of the delta (flooded daily to fortnightly by tides)		
Beaches	Tidal sandy slope	Moderate gradient sloping sandy surface subject to wave action and flooded daily to fortnightly by tides
Intertidal sand flats	Tidal sand flat	Low gradient sloping flat underlain by sand and flooded daily to fortnightly by tides
Intertidal mud flats	Tidal mud flat	Low gradient sloping flat underlain by mud and flooded daily to fortnightly by tides
Swale basins leeward of sand bars and beach ridges and inundated by tides	Tidal lagoons	Basins formed leeward of a sand bar or beach ridge and whose floors are sandy, terrigenous mud, or carbonate mud; the basins are inundated by tides
Tidal creek	Channel	Channel, floored by mud or sand and flooded daily to fortnightly by tides
Spit or chenier	Low relief emergent landform	Convex surface underlain by sand or shell and sand, usually flooded by the highest tides; also inundated by high river floods

been built up beyond the level of the high tide to levels of high floodwaters by accretion of sediment from river-flood events. In the latter situation, they are subject to prevailing floodwaters that also can erosionally plane the delta surface. Nearly all landforms within a delta are wet by one or more of the processes listed above. As such, the majority of landforms within the delta are "wetlands," but rather than viewing the delta as a single wetland, the delta actually is an integrated wetland complex composed of a mosaic of wetland types. Each of the wetlands within this delta complex may have been developed by different sedimentologic/geomorphic processes (e.g., a shoaled tidal flat vs. a vertically accreted floodplain or a lagoon behind a deltaic wave-built barrier/bar acting as a lake vs. a water-filled abandoned distributary channel or water-filled abandoned meander), and each wetland type may be maintained by one or more of the hydrologic mechanisms of maintenance listed above. Each of the delta types, whether fluvial, wave, or tide dominated, tends to have their own characteristic suite of wetlands (Figure 4). Examples of deltas with various wetlands therein are illustrated in Figure 5. Sedimentary and vegetation descriptors have been added to wetland categories. The range of wetlands commonly occurring in estuarine deltas is listed and described in Table 2.

The hydrology and hydrochemistry of a deltaic system, once it has aggraded/prograded and become emergent to near or above the level of high tide, generally are complex. This is particularly so for large deltas that comprise a suite of landforms and stratigraphic units, a range of water sources of different hydrochemistry, and a range of



Estuarine Deltaic Wetlands, Figure 6 Diagrammatic transverse (shore-parallel) cross section of an aggraded delta showing main features of delta hydrology and hydrochemistry and hydrologic interactions.

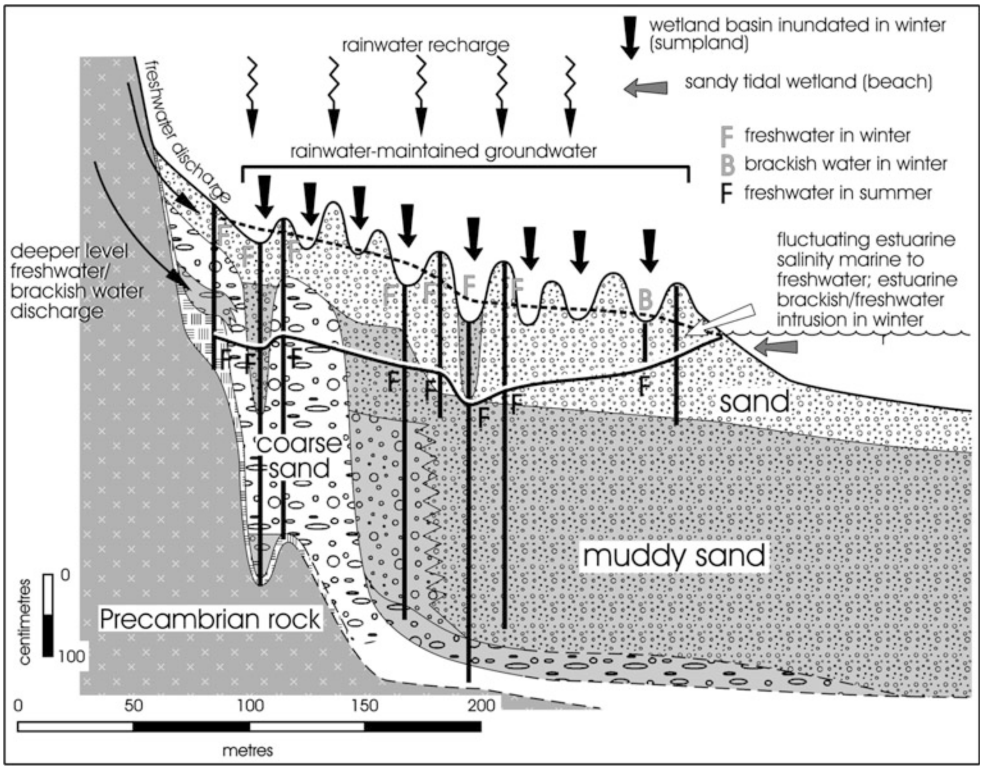
hydrologic recharge mechanisms (viz., rain-recharged, tide-recharged marine/brackish water, tidally driven freshwater, seasonally rising and falling water table above freshwater lenses, and seasonally rising and falling water table above saline water lenses, the latter varying seasonally). Deltas set in the various possible tidal ranges and climate types provide further variation on the dynamics of hydrology and hydrochemistry.

In the midst of standard salinity gradients and intra-annual variations resulting from seasonal inflows and the size of the estuary, there are important localized interactions occurring between water bodies. While surrounded by open estuarine waters (into which the delta has been prograding) and distributary channel systems that can vary from marine to brackish to freshwater seasonally, the core of a delta may contain a freshwater lens (similar to the freshwater lens under an oceanic island) that interacts hydrologically on its margins with the estuarine water body or the channels. As such, the contact of the freshwater lens has a saline/freshwater contact similar to the Ghyben-Herzberg saline and freshwater relation (Todd, 1959). Local areas on the subaerial delta plain, comprising contrasting mud-floored lagoons/ponds or sandy cheniers, or abandoned channels, are affected by river inflow, or rain, or evaporation and develop salinities and hydrochemically specific surface and near-surface water bodies and groundwater bodies, respectively, that perturbate the salinity/hydrochemistry of the main body of the delta groundwater. A summary of the main features of delta hydrology and hydrochemistry and hydrologic interactions is shown in Figure 6. Details of the hydrologic and hydrochemical features and dynamics in beach ridges and wetland swales of a wave-dominated delta in Walpole-Nornalup Inlet Estuary of southwestern Australia are illustrated in Figure 7. This latter case study illustrates the changing water tables and salinity in groundwater and estuarine water between wet and dry seasons in the delta; the dynamics of discharge, intrusion, and seepage that occur between the delta groundwater and estuarine water; and the wetlands that occur on this wave-dominated estuarine delta. The various water bodies and their hydrodynamic characteristics and interactions as shown in Figures 6 and 7 are underlying determinants of the biota resident on the deltaic wetlands.

Wetlands on deltas, depending on their origin, height relative to a water table, and location relative to the river source or marine outlet, have specific characteristics stratigraphically, pedogenically, hydrologically, and hydrochemically. As such, they may function as different habitats for biota.

Biota inhabiting the wetlands of estuarine deltas

A wide range of biota inhabit deltaic estuarine wetlands and the biotic assemblages thereon can be quite variable and complex because the wetlands divide in the first



Estuarine Deltaic Wetlands, Figure 7 Hydrologic and salinity characteristics and dynamics in beach ridges and wetland swales in a section longitudinally through the wave-dominated Deep River Delta in Walpole-Nornalup Inlet Estuary of southwestern Australia (information from Semeniuk et al., 2011).

instance into terrestrial (fluvial) types and coastal (marine/estuarine) types and in the second instance the type of biota that inhabits these environments is determined by a number of factors: biogeography and climate, substrates, salinity and hydrochemistry, hydroperiod (frequency of flooding, inundation, or wetting), height of the wetland above MSL, and tidal range. For instance, in tropical climates mangroves colonize the mid- to high-tidal zones (habitats) of deltas, while in temperate climates there is salt marsh in this zone. On low-tidal flats, there is often an invertebrate benthos of varying biodiversity related to biogeography and climate. Low-tidal flats comprising sand in tropical climates support a range and diversity of invertebrates that include molluscs, crustaceans, echinoderms, polychaetes, and anemones, while those in temperate climates support a lower diversity of molluscs, crustaceans, and polychaetes. Low muddy tidal flats in tropical climates support a more diverse range of molluscs, crustaceans, and polychaetes, than those in temperate climates. A description of some of the typical biota that inhabits estuarine deltaic wetlands in Australia as related to climate, hydrography/oceanography, tidal range, tidal level, wetland type (habitat), substrate, hydroperiod, and salinity is provided in Table 3.

Examples of some of the biotic assemblages on deltaic wetlands in the Lawley River Estuary, Leschenault Inlet Estuary, and Swan-Canning Estuary are shown in Figure 8. The muddy tide-dominated macrotidal delta of the Lawley River Estuary supports mangroves on mid- to high-tidal flats, salt marsh on the high-tidal alluvial fans, and high-diversity molluscs, crustaceans, and polychaetes in the low-tidal flats. The sand-and-mud tide-dominated microtidal Preston River Delta of the Leschenault Inlet Estuary supports mangroves and salt marsh in the mid- to high-tidal flats and molluscs, crustaceans, and polychaetes in the low-tidal flats. The sand-and-mud tide-dominated microtidal delta of the Swan-Canning River Estuary supports salt marsh and forests/heath in the mid- to high-tidal flats and molluscs, crustaceans, and polychaetes in the low-tidal flats. For the Canning Delta of the Swan-Canning River Estuary, the vegetation manifests a plethora of floristic units within the salt marsh, compared to the salt marsh of the Lawley River Delta and the Preston River Delta, and so it presented in more detail to illustrate the complexity of floristic units that can occur on such estuarine deltas.

Discussion and summary

The landforms that comprise the deltas are the foundation to wetlands and the habitats for biota. The landform types and hence the wetland types vary according to whether the delta is fluvially dominated, wave dominated, or tide dominated.

Estuarine Deltaic Wetlands, Table 3 Examples of deltaic wetland in estuaries in Australia and their biota

Estuary and climate	Delta type	Delta wetland habitats	Hydrography/ oceanography	Substrate	Hydroperiod/ salinity	Biota
Lawley River estuary, tropical humid	Tide-dominated delta	Supratidal floodplain and anastomosing shallow river channel	River-dominated, terrestrial	Sand and mud on floodplain, gravel and sand in channel	Flooded annually in the wet season by freshwater	Salt marsh on the floodplain, paperbarks, and salt marsh fringing the channel
		High-tidal to mid-tidal flats	Tide-dominated, macrotidal	Mud	Tidally flooded daily to fortnightly, salinity ~ 40 ppt grading upslope to 190 ppt	Moderately diverse zoned mangroves and accompanying high-diversity benthic fauna between MSL and MHWS, samphire at ~ MHWS, bare salt flat with local algal mat at levels above MHWS with crustacean and insects
		Low-tidal to mid-tidal flats	Tide-dominated, macrotidal	Mud	Flooded daily, salinity ~ 35 ppt	High-diversity infauna of crustacean, molluscs, polychaetes, sponges
Peel-Harvey Estuary, Harvey River delta, subtropical subhumid	Fluvially dominated delta	Supratidal floodplain, abandoned channels (sumplands), cheniers, and shallow river channel	River-dominated, terrestrial	Sand and mud on floodplain, gravel and sand in channel	Flooded annually in the wet season by freshwater	Salt marsh on the floodplain; paperbarks forest, heath, and salt marsh fringing the channel on the levees; salt marsh on the interior floodplains
		Supratidal levee	River-dominated, terrestrial	Sand and mud	Flooded annually in the wet season by freshwater otherwise wetted by brackish shallow groundwater	Paperbark forest and heath
		Supratidal lagoons/ ponds (sumplands)	River-dominated, terrestrial	Sand and mud on floodplain, gravel and sand in channel	Flooded annually in the wet season by freshwater	Fringing salt marsh, local algal mats, gastropods
		Tidal flats in front of delta	Fluvially dominated, microtidal	Sand	Tidally flooded daily, salinity ~ 40 ppt	Low diversity invertebrate infauna of crustacean, molluscs, and polychaetes
		Tidal flats lateral to delta	Fluvially dominated, microtidal	Mud	Tidally flooded daily, salinity ~ 40 ppt	Low diversity invertebrate infauna of crustacean, molluscs, and polychaetes

Estuarine Deltaic Wetlands, Table 3 (Continued)

Estuary and climate	Delta type	Delta wetland habitats	Hydrography/ oceanography	Substrate	Hydroperiod/ salinity	Biota
Walpole-Nornalup Inlet Estuary; Deep River Delta; temperate, humid	Wave-dominated delta	Swales between beach ridges (sumplands) on the supratidal delta plain, dissected by distributary channels	Wave-dominated, microtidal	Sand, peaty sand, and peat	Recharged annually by rain and water table rise, freshwater	Vegetated by wetland forests, heath, and rushes/sedges; sparse invertebrate infauna of crustacean, insects
		Lagoons (lakes and sumplands) on the supratidal delta plain	Wave-dominated, microtidal	Sand, peaty sand, and peat	Recharged annually by rain and water table rise, freshwater	Vegetated by wetland heath, and rushes/ sedges; sparse infauna of crustacean, insects
		Tidal flats seaward of subaerial delta	Wave-dominated, microtidal	Sand	Flooded daily by tides and annually by river, marine/ brackish salinity varying to freshwater annually	Sparse infauna and low diversity of molluscs, crustacean, polychaetes
Swan-Canning Estuary; subtropical, subhumid	Fluvially dominated delta	Subaerial shoals oriented parallel to river flow (floodplains), abandoned channels (sumplands), and dissected by distributary channels	Fluvially dominated, microtidal	Sand, peaty sand	Recharged annually by rain, water table rise, and river flooding; freshwater	Vegetated by wetland forests, heath, and rushes/sedges; sparse infauna of crustacean, insects
		Tidal creeks dissecting the subaerial delta	Fluvially dominated, microtidal	Sand and mud	Flooded daily by tides, flooded annually by river; marine/ brackish salinity	Margins vegetated by salt marsh (zoned chenopods and sedges/rushes), sparse infauna of low diversity molluscs, crustacean, polychaetes, and sea grass
		Tidal flats seaward of subaerial delta	Fluvially dominated, microtidal	Sand and mud	Flooded daily by tides, flooded annually by river; marine/ brackish salinity	Sparse infauna, low diversity of molluscs, crustacean, polychaetes, and sea grass
Leschenault Inlet Estuary; Collie River delta; subtropical, subhumid	Mixed wave and fluvially dominated delta	Subaerial plain (floodplains) with cheniers, abandoned channels (sumplands), and dissected by distributary channels	Fluvially and wave-dominated, microtidal	Sand, peaty sand	Recharged annually by rain, water table rise, and river flooding; freshwater	Vegetated by wetland forests, heath, and rushes/sedges; infauna of crustacean, insects

Estuarine Deltaic Wetlands, Table 3 (Continued)

Estuary and climate	Delta type	Delta wetland habitats	Hydrography/ oceanography	Substrate	Hydroperiod/ salinity	Biota
		Tidal flats seaward of subaerial delta	Mixed fluvially- and wave-dominated, microtidal	Sand and mud	Flooded daily by tides, flooded annually by river; marine/ brackish salinity	Both sand-and-mud tidal flats inhabited by sparse infauna of low diversity molluscs, crustacean, polychaetes, and sea grass
Westernport Bay, Bass River delta; temperate, humid	Fluvial-dominated delta	High-tidal flats	Fluvially dominated, microtidal	Mud		Mangroves and salt marsh zoned across tidal flat
		Low-tidal flats	Fluvially dominated, microtidal	Mud		Sea grass beds, high diversity of benthic invertebrate fauna (molluscs, crustacean, polychaetes)
Hawkesbury River estuary, bayhead delta; temperate, humid	Fluvial-dominated delta	Mid- to high-tidal flats	Fluvially dominated, microtidal	Mud		Mangroves and salt marsh zoned across tidal flat
		Low-tidal flats	Fluvially dominated, microtidal	Mud		Sea grass beds, molluscs, crustacean, polychaete fauna

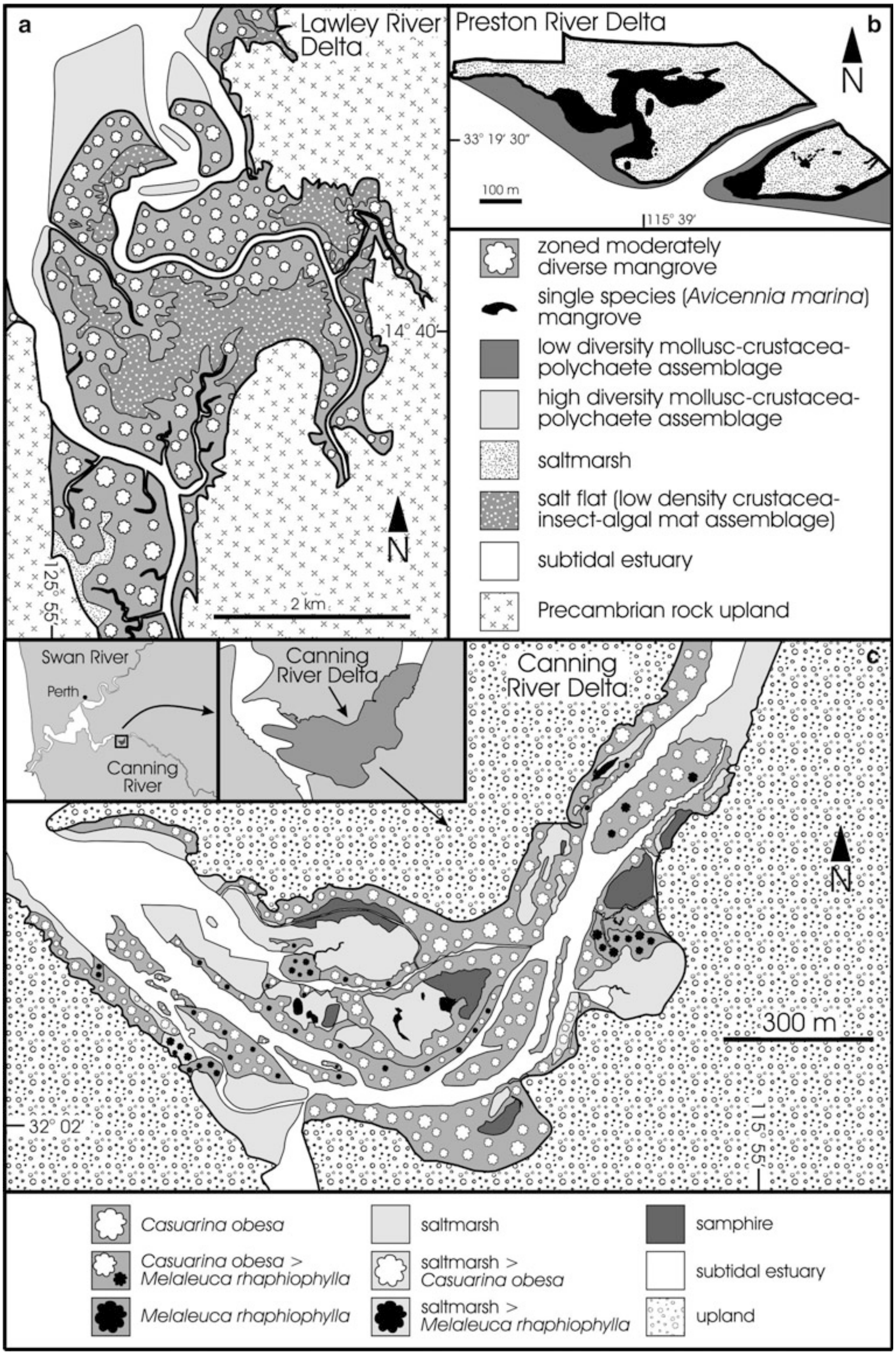
Information collated from Semeniuk and Semeniuk (1990), Saintilan and Hashimoto (1999), Semeniuk et al. (2000, 2011), Pen et al. (2000), Semeniuk (1983, 1985, 2000), Department of Sustainability and Environment (2003), Brearley (2005)

A recurring suite of wet landforms on deltas are the tidally wetted tidal flats, tidal creeks, and tidal shoals and the subaerial parts of deltas wet by rain, floodwater, and groundwater such as floodplains, local cheniers, distributary channels, levee banks, water-filled and sediment-filled abandoned channels, swales, and lagoons. The terrain of a delta is essentially a nearly flat, low-lying surface close to a water table, and, as such, the majority of landforms within the delta are "wetlands," but rather than viewing the delta as a single wetland, the delta actually is an integrated wetland complex composed of a mosaic of different wetland types. Each may have been developed by different sedimentologic/geomorphic processes, and each may be maintained by one or more of the hydrologic mechanisms operating on the delta. The hydrology and hydrochemistry of a delta system generally is complex, particularly for large deltas where there may be a large number of landforms and stratigraphic units, a range of water sources of different hydrochemistry, and a range of hydrologic recharge mechanisms. Dependent on whether the wetting of the terrain is by freshwater (rain or river water), estuarine water, or marine water and on how frequent is the wetting process, the groundwater and/or surface water of the wetland may be freshwater, brackish, or saline, and it may be stasohaline or poikilohaline.

As such, a large range of wetlands can be developed on a delta within an estuary, and there are a large range of possible habitats for biota, determined by landforms, substrates, hydrology, hydroperiod, and salinity/ hydrochemistry.

The subaerial wetlands are classified according to their geomorphic and hydrologic characteristics (using the "geomorphic-hydrologic classification") and geomorphically for tidal wetlands. The "geomorphic-hydrologic classification," recognizes that deltaic wetlands are mainly basins, flats, and channels, and the hydrological regimes that maintain them are permanent inundation, seasonal inundation, seasonal waterlogging, and intermittent inundation. For the tidally inundated part of the deltas, the wetlands are identified as to geomorphic units, viz., tidal flats, tidal creeks, and spits and cheniers.

Wetlands on deltas, depending on their origin, height relative to a water table, and location relative to the river source or marine outlet, have specific characteristics stratigraphically, pedogenically, hydrologically, and hydrochemically. As such, they may function as different habitats for biota. A wide range of biota inhabits deltaic estuarine wetlands developed on terrestrial wetland types and marine/estuarine wetland types and determined by substrates, salinity and hydrochemistry, hydroperiod, height of the wetland above MSL, and tidal range.



Estuarine Deltaic Wetlands, Figure 8 Biota inhabiting various estuarine deltaic wetlands. (**a**) The tropical macrotidal muddy tide-dominated delta of the Lawley River Estuary with its moderately diverse zoned mangroves (species of mangrove listed in Semeniuk, 1983, 1985). The salt marsh species are listed in more detail in Cresswell et al. (2011). (**b**) The subtropical microtidal sand-and-mud tide-dominated delta of the Preston River Delta of the Leschenault Inlet Estuary with its single species mangrove (*Avicennia marina*); the salt marsh species are listed in more detail in Pen et al. (2000). (**c**) The subtropical microtidal sand-and-mud fluvial-dominated Canning River Delta of the Swan-Canning River Estuary. The salt marsh and samphire species are listed in more detail in Pen (1983).

The type of biota inhabiting these environments is also determined by biogeography and climate. Mangroves are common in the mid- to high-tidal zones of deltas in tropical climates and salt marshes occupy similar habitats in temperate climates. Low-tidal flats generally support an invertebrate benthos of varying biodiversity that include molluscs, crustaceans, echinoderms, polychaetes, and anemones, as related to biogeography and climate.

Estuarine deltas are a nexus for marine, tidal, and terrestrial wetlands to come together. Although each category of wetland is subject to its own developmental and functional sedimentary, hydrological, and hydrochemical processes, they combine to illustrate the river-to-marine transitional environment of deltas and conformably fit the overall geometry of delta types (tide or wave dominated). Whether they have a voluminous or scant sedimentary budget, the delta will still contain a variety of wetland types specific to its environmental conditions.

While each delta comprises a suite of small-scale landforms which are subject to wetting processes, from diurnal to seasonal to intermittent, and the patterns of inundation, waterlogging, and evaporation determine their hydrochemistry and salinity, onto this pattern is projected the effect of small-scale sedimentary lenses which can store and discharge water at different rates and through different pathways. These small-scale hydrological responses create interactions with their surrounding bodies of water altering the hydrochemistry and salinity along edges, margins, and juxtapositions. As a result of these interactions, deltaic wetlands may differ not only in geometry and wetting mechanisms but, in the case of similar wetland types, importantly, for biota, in hydrological and hydrochemical characteristics.

Small-scale wetlands within the deltaic complex provide a greater range of habitats than can be seen from a simple map, as is apparent when comparing the examples of the various deltaic wetlands. It is also clear that biota discriminate at this localized and small-scale level.

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Cross-references

Coastal Wetlands
Delta Plain
Deltas
Wetlands

ESTUARINE FLOCCULATION

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Synonyms

Aggregation; Coagulation; Floc formation

Definition

Flocculation is a process by which particles come out of the suspension to form flocs or larger particles, specifically in an estuarine environment.

The flocculation process is influenced by physical, chemical, and biological factors in both terrestrial and aquatic environments. Flood, ebb, and wave energies drive the suspension and resuspension of sediments and affect the formation and differential settlement of flocs. The concentration of dissolved ions and particulate and dissolved organic matter influences floc surface charge, promoting flocculation as well. These factors, together with estuarine circulation patterns, provide an important mechanism for suspended particles to flocculate and later settle to the estuarine floor (Eisma, 1986; Day et al., 1989; McLusky and Elliott, 2004; Wolanski, 2007).

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Cross-references

Sediment Grain Size
Wave-Driven Sediment Resuspension

ESTUARINE GEOMORPHOLOGY

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Definition

Estuarine geomorphology is the study of the landforms of an estuary. Geomorphologically, an estuary is defined as a semi-enclosed body of water which contains landforms derived from both fluvial and marine processes, where the latter contain sediment that has been transported landward by wind, wave, and tidal processes.

Description

An estuary is the seaward portion of a drowned valley system (Dalrymple et al., 1992) and is divided into wave- and tide-dominated forms (Kennedy, 2011). The geomorphology of a wave-dominated estuary is subdivided into three zones. The marine zone occurs at the oceanward end and is where beach-barrier systems as well as flood and ebb tide deltas are found. The central part of the estuary is a low-energy zone dominated by a mud basin, while at the landward end, river deltas occur in the fluvially dominated zone (Roy, 1984). In tide-dominated estuaries, barrier systems are absent, and the marine zone is composed of a series of elongate subtidal bars extending far into the estuary. The central mud basin does not occur (Dalrymple et al., 1992; Dalrymple and Choi, 2007). In planform these estuaries are funnel shaped, being characterized by straight marine and fluvial channels at opposite ends of the estuary separated by a meandering channel zone where tidal and river flows interact (Woodroffe et al., 1993; Dalrymple and Choi, 2007).

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Cross-references

Coastal Landforms
Estuarine Geomorphology
Geomorphological Mapping

ESTUARINE HABITAT RESTORATION

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Synonyms

Conservation; Creation; Enhancement; Habitat reclamation; Mitigation; Protection; Rehabilitation

Definitions

Restoration means the return of an ecosystem to a close approximation of its previously existing condition (e.g., NRC, 1992; Lewis, 2000). Habitat restoration is defined as returning the condition of an altered or disturbed habitat to a close approximation of a former (less disturbed) or pre-disturbance condition. In the common vernacular, however, restoration appears to be an umbrella term encompassing creation, enhancement, rehabilitation, reclamation, and even mitigation. Yet, each of these terms has distinctly different meanings as defined in the literature. Unless otherwise specified, the term *restoration* refers to any form of human intervention with the intent of improving upon the existing condition of an ecosystem or habitat.

The term "habitat" generally refers to a more or less discrete biophysically structured element of an ecosystem. A habitat can also be defined as the portion of an ecosystem where a species lives, feeds, and reproduces. Commonly referred to estuarine habitat types include tidal forested swamps, shrub scrub wetlands, emergent marshes, seagrass meadows, seaweed beds, and unvegetated channels and flats (Cowardin et al., 1979). An estuarine ecosystem generally includes a complex of several habitat types that interact within a geomorphic setting along the estuarine gradient.

Introduction

Restoring estuarine habitats generally means repairing damages caused by humans and natural forces. Because of the extensive human occupation, development, and use of coastal areas for centuries, the extensive estuarine habitats have been either destroyed or significantly impaired.

Estuarine habitat alteration

Of the prehistoric area of natural habitats in estuaries, a large but poorly quantified portion has either been destroyed or highly altered (NRC, 1992). In the United States, much of this change occurred from the late 1800s with the development of ports, commerce, and agriculture. Alterations and losses continued at a rapid pace until approximately the late- 1960s with the advent of environmental laws (e.g., US National Environmental Policy Act). Loss of estuarine habitat is common on most major continents (MEA, 2005). Estuaries in Europe showed alterations and loss. Extensive estuarine modifications in East Asia have been ongoing for millennia but estuaries have suffered rapid losses over the past several decades because of development actions. Estuaries in the Middle East, Africa, and Australia have also suffered habitat losses.

In general, coastal and estuarine habitats have undergone rapid losses in most regions of the world, largely because of human development in coastal areas. The worldwide decline in seagrasses has been termed a "global crisis" (Orth et al., 2006). Similar degrees of losses have been estimated for mangrove and tidal swamps (Mitch and Gosselink, 1993). Tidal marshes may be the most documented estuarine habitats in terms of ecology and changes. Data show that the total tidal marsh area in the United States has decreased from an estimated 2,023,000 ha in 1950 to about 1,850,000 ha in 2004 (Dahl, 2006). Currently, the greatest rates of tidal marsh loss exist in the Gulf of Mexico region where it is estimated that approximately 2,331 ha are lost per year (Dahl, 2006). Between 1986 and 2004, the United States lost 0.9 % of estuarine emergent wetlands on an annual basis (Dahl, 2006). Many bays and estuaries have lost more than 90 % of their tidal wetland area, especially in urban areas, and many others are in a state of poor or declining health (Borde et al., in preparation). Human alterations often associated with urban development, such as the construction of infrastructure, can have permanent and irreversible effects on tidal marshes. Other human-related alterations, such as dredging, filling, ditching, diking, chemical contamination, and hydrologic changes, are potentially reversible with restoration. Natural stressors to estuarine habitats include climate variability, catastrophic storm events, sea-level rise, unexplained marsh dieback, and invasive species.

Advent of estuarine habitat restoration

Restoration of tidal wetlands in the United States has one of the longest histories among all restoration efforts (NRC, 1992). Planting of *Spartina* marshes in the eastern United States commenced in earnest during the 1970s and demonstrated that marshes could be created and restored. With the development of wetland regulations as part of the US Clean Water Act, wetland mitigation projects were designed to compensate for damages associated with development, which triggered development of tidal wetland restoration science. Over a century ago, Ducks Unlimited and other organizations developed strategies to conserve and restore tidal wetlands. In 1990, the National Oceanic and Atmospheric Administration (NOAA) established its Office of Restoration and published a book of case studies for restoration of marine habitats that included tidal wetlands (Thayer, 1992). One encouraging strategy is the focus on community-based projects that have strong local support. The NOAA conducts workshops and publishes guidance documents (Thayer, 1992; Hackney, 2000; Thayer et al., 2003; Thayer and Kentula, 2005) to foster the underlying science.

The National Research Council (1992) recommended to the Congress that any redirection of federal policies and programs for aquatic ecosystem restoration should consider the following:

- The use of a landscape perspective in restoration efforts
- The use of adaptive planning and management
- Evaluating and ranking restoration alternatives based on assessment of opportunity–cost rather than on traditional benefit–cost analysis
- Incorporating the definition of restoration as the return of an ecosystem to a close approximation of its condition prior to disturbance
- Reliance on nonfederal and federal units of government to coordinate restoration programs in local areas
- Initiating an interagency and intergovernmental process to develop a unified strategy for aquatic ecosystem restoration

Aspects of each of these recommendations have been implemented by nongovernmental organizations such as Restore America's Estuaries (RAE and NOAA, 2002); however, the implementation of an integrated strategy remains a work in progress. The US Army Corps of Engineers has identified ecosystem restoration as one of its three primary missions and has the regulatory role and technical expertise to make a significant contribution to a national strategy (USACE, 2007a).

The realization that estuarine habitats provide significant ecosystem services (i.e., provisioning, regulating, cultural, and supporting benefits to people) has reinforced global efforts to protect, conserve, and restore these habitats (MEA, 2005). It has been long understood that estuarine habitats support fisheries resources. Many habitats produce vast amounts of organic matter that support the food web and serve as feeding, rearing, and refuge habitat for many fisheries species. Further, coastal and estuarine habitats have contributed to the resilience of coastlines to withstand and recover from storm surge and other natural disturbances that could harm coastal communities, coastal resources, and infrastructure. Wetlands are well known for their ability to trap suspended sediments, process organic matter and inorganic nutrients, and attenuate floods. Estuarine and near-coastal shallow-water habitats can be major sinks of carbon dioxide, thus capable of mitigating anthropogenic carbon dioxide emissions contributing to global warming and ocean acidification (MEA, 2005). Therefore, restoring habitats to restore ecosystem services has become another goal of programs in many countries.

Elements of successful estuarine habitat restoration

The establishment of clearly articulated restoration goals and selection of the right restoration strategy to meet those goals are critical to successful habitat restoration. The results of actions taken on habitat restoration sites increasingly inform our understanding of ecosystems and enhance our ability to accurately predict how well-planned actions will meet restoration goals. Successful estuarine habitat restoration considers the characteristics and processes of the overall landscape within which the restoration site is found, as well as an array of restoration actions. Solid project management from planning through implementation and project evaluation ensures success; restoration projects typically involve well-conceived monitoring, data management, dissemination of results, and adaptive management to continuously improve processes and understanding. The use of a variety of models helps prioritize projects, develop performance criteria from goals, improve project design, assist with site selection, etc. Each restoration effort aims to ensure long-term habitat viability and stability in the face of disturbances often relative to reference (control) sites that represent the targeted natural condition. Beyond the costs of habitat revegetation, restoration involves the costs of site assessments, site acquisition, site preparation, safety considerations, permitting, and project management.

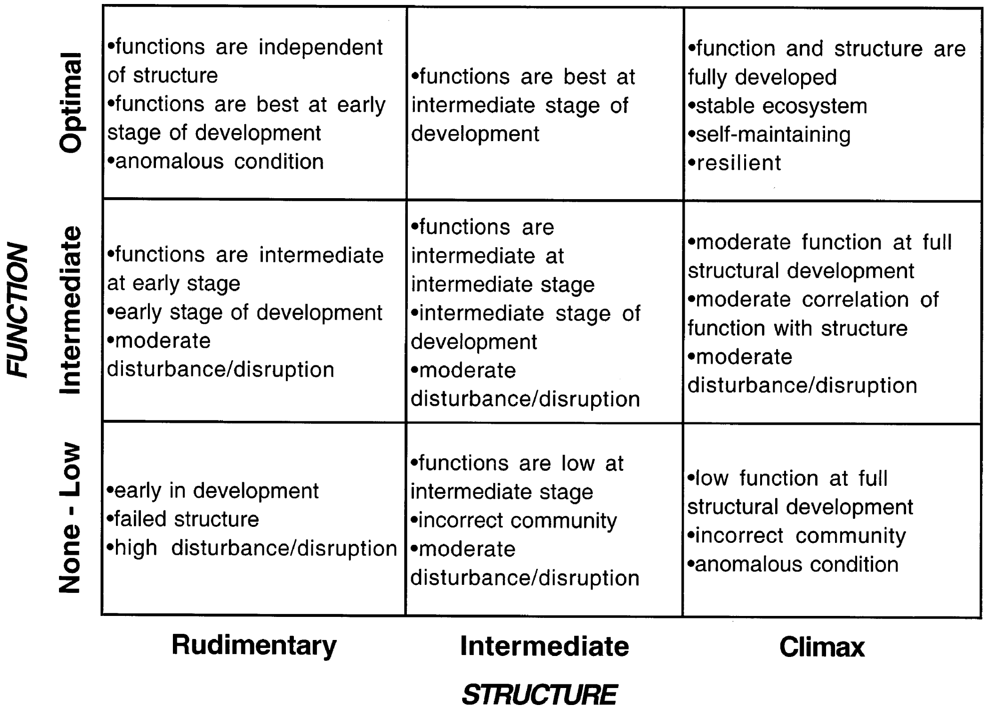
Defining restoration goals

The general intent of tidal marsh restoration is a net improvement of the ecological functions and services of the marsh. This suggests that upon completion of a restorative action, a marsh will exhibit quantified improvement in size, structural features (e.g., vegetative cover, species richness), and function (e.g., productivity, use by wildlife). Practitioners have learned that satisfying this intent generally requires far more than simply installing marsh plants at a site.

The effect an ecosystem has on restoring habitats is exemplified by how tidal wetlands depend upon processes in their landscape for support. For example, the accretion of the marsh plain depends on a supply of inorganic sediment from the surrounding upland. If the sediment is not available, marshes tend to subside (i.e., sink), which forces changes in their structure and function. Ideally, the landscape provides the sediment, hydrology, and nutrients that form and help sustain the tidal wetland. It is well documented that the probability of successful restoration is largely dependent on the degrees of site and landscape disturbance (NRC, 1992). Consideration of the health of landscape processes is essential in the planning phase of a site-specific restoration project.

Goals for a habitat restoration project must be specific enough for the proper restoration strategies and actions to be identified and implemented. Goals can be defined in terms of the structural (i.e., species composition and abundance) or functional (e.g., productivity, nutrient processing, organic matter production) *state* that the project site will reach after implementation of restorative actions. The theory behind ecosystem-based approaches to aquatic ecosystem restoration was outlined by the NRC (1992). A general model for ecosystem state (Figure 1) is a way to visualize the present and historical *states* of the system, as well as identify restoration goals. This model assumes there is a positive relationship between the structure and the function of an ecosystem. To simplify the model, the axes can be divided into low, moderate, and high ecosystem functioning to define nine system states. Dividing the matrix into three levels (1) acknowledges the uncertainty about the relationship between structural and functional ecosystem components and (2) recognizes the natural dynamic variability associated with structural conditions and functional conditions within a state. The natural climax structure (upper right-hand box) of an ecosystem, habitat, or community has a corresponding and predictable functional condition. The targeted end state, then, would represent a system that is fully developed, optimally biodiverse, self-maintaining, and resilient enough to withstand and recover from disturbances.

For example, the upper right-hand box labeled desired ecosystem state might represent the pre-disturbance conditions of the lower Columbia River and estuary (Johnson et al., 2003). It can also represent the desired state of the system after restoration. The present condition



FUNCTION	Rudimentary	Intermediate	Climax
Optimal	•functions are independent of structure •functions are best at early stage of development •anomalous condition	•functions are best at intermediate stage of development	•function and structure are fully developed •stable ecosystem •self-maintaining •resilient
Intermediate	•functions are intermediate at early stage •early stage of development •moderate disturbance/disruption	•functions are intermediate at intermediate stage •intermediate stage of development •moderate disturbance/disruption	•moderate function at full structural development •moderate correlation of function with structure •moderate disturbance/disruption
None - Low	•early in development •failed structure •high disturbance/disruption	•functions are low at intermediate stage •incorrect community •moderate disturbance/disruption	•low function at full structural development •incorrect community •anomalous condition

STRUCTURE

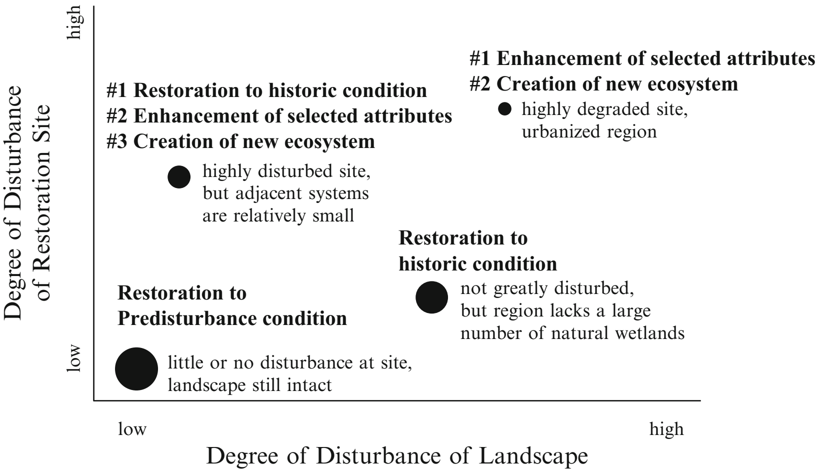
Estuarine Habitat Restoration, Figure 1 Restoration system development matrix (from Thom, 2000) showing development of a system from early rudimentary stages to a fully developed system structurally and functionally.

of the system is altered from prehistoric conditions and is not in a desirable state. The structure and functioning of the system is different than it was prior to hydrologic modification and other changes, and it probably has reached equilibrium in this altered state. The growing body of information indicates that the survival of salmon may be dependent on the return of the estuary to a less altered state. With regard to salmon that as juveniles use the floodplain habitats, the system now exists in what can be termed an unacceptable ecosystem condition. Under a restoration scenario, if the system occupies any set of values that are within the state identified as desirable ecosystem condition, then it has met the goal. Because of uncertainties related to natural variability, influences from the surrounding landscape, and low predictive capabilities, we can reliably only get close to the goal (Shreffler and Thom, 1993; Hobbs and Norton, 1996). Also, in a case where the habitat is not in the desirable state, there should be plausible explanations. From these explanations, options for actions to improve conditions can be recommended (Thom, 1997, 2000).

Research on estuarine habitat restoration has focused chiefly on techniques of species establishment and on development of community structure (NRC, 1992; Warren et al., 2002; Moreno-Mateos et al., 2012). The functions of estuarine habitats, although widely recognized, are seldom evaluated during post-restoration monitoring. The science of estuarine habitat restoration and creation has not yet evolved to the point that lost habitat functions can be predictably replaced (Kusler and Kentula, 1990). Further, the feasibility of restoring or creating habitats has been demonstrated, but their functions, stability, and resiliency are largely unknown (Thayer, 1992). Zedler and Weller (1990) concluded that "It has not been shown that restored or constructed wetlands maintain regional biodiversity and recreate functional ecosystems – there is some evidence that constructed wetlands can *look* like natural ones; there are few data to show they *behave* like natural ones."

Restoration strategies

Restoration of natural systems in general can involve several strategies (Thom et al., 2005). The application of these strategies is dictated by which one is the best solution to the problem, the probability that the strategy will work to produce the desired outcomes, the period of time to reach desired outcomes, feasibility, and cost, etc. Returning a habitat to pre-disturbance conditions is difficult in many cases and almost impossible where the habitat and the landscape in which it occurs are both highly disturbed and altered. Hence, restoring habitat to some alternative state, referred to as a *novel* habitat or ecosystem, may be the only feasible outcome (Thom, 1997). Novel habitats or ecosystems are characterized by structural and functional conditions of partial recovery to a pre-disturbance state (Figure 1).



Estuarine Habitat Restoration, Figure 2 Level of disturbance on site and landscape scales and restoration strategies that have the greatest chance of working under these conditions. (From Thom et al., 2005).

The restoration strategies (Figure 2) include the following:

- *No Intervention* – Under this strategy, recovery is left entirely to natural processes. The outcome is unpredictable and may not resemble pre-disturbance conditions. Two possible trajectories of the no-intervention approach are natural recovery and further degradation. Natural recovery of severely disturbed or altered habitats can take decades to centuries. Further degradation may be an alternative steady state in the progression toward natural recovery.
- *Creation of a New Ecosystem* – This strategy involves bringing into being a new ecosystem that previously did not exist on the site (NRC, 1992). The goal is to emulate the present condition of an existing, functioning reference ecosystem. Ecosystem creation can involve elaborate reconstruction of both physical (e.g., topographic, hydrologic) and biotic (e.g., native plants) elements. Although created ecosystems may eventually become self-maintaining, there can be considerable uncertainty about the quality and sustainability of the habitat. Created ecosystems typically require substantial ongoing management.
- *Enhancement* – This strategy involves the intentional alteration of an existing habitat to provide conditions that previously did not exist and which by consensus increase one or more attributes (NRC, 1992). Enhancement and restoration are often confused (Lewis, 2000). For estuarine systems, enhancement means *enhancement of selected attributes* of the ecosystem such as improving the quality, diversity, or size of an existing marsh or seagrass meadow. Enhancement, as opposed to true restoration, may represent the most often-applied strategy.
- *Restoration* – Restoration means the return of an ecosystem to a close approximation of its previously existing condition (e.g., Lewis, 2000; NRC, 1992). It involves taking an action to increase the rate of recovery over the rate of natural recovery that would occur without human intervention.
- *Conservation* – Conservation generally refers to the maintenance of biodiversity. Under this strategy, some development can occur as long as biodiversity and the structure and processes to maintain it are not affected.
- *Protection* – Protection involves exclusion of activities that may negatively affect the structure and/or functioning of habitats or ecosystems. It can also involve protection of a species through management actions that eliminate harm to a species directly or indirectly through damage to its habitat. Marine-protected areas are an example of this strategy.

Selecting the right strategy

Among the first steps in habitat restoration is to determine what is “wrong” with the existing system. After clearly articulating the problem, the next step is to arrive at possible solutions to the problem. Then the appropriate strategy or combination of strategies is selected. Strategies are often used in combination. For example, enhancement of a damaged seagrass meadow through transplanting can be done in combination with fisheries conservation measures. Once the factors causing the damage are abated, seagrasses can be planted. In conjunction with plantings, conservation measures, which limit the catch of seagrass-dependent species, can be invoked. The additional habitat along with the reduced fishing pressure result in a net increase in species populations, and they contribute to the well-being of the other seagrass-associated functions and species.

Using reference sites

Reference systems are habitats or a matrix of habitats that are generally minimally disturbed and represent the natural model that a restoration project is designed to achieve. They serve three primary functions: (1) they can be used as models for developing restoration actions; (2) they provide a target from which performance goals can be derived and against which progress toward these goals can be compared; and (3) they provide a comparison system by which environmental fluctuations unrelated to the restoration action can be assessed. Alternatively, degraded control sites can be used to show the progress of the restored system away from the degraded condition (NRC, 1992).

Horner and Radaeke (1989) identified the following features that should be assessed for degree of similarity between a reference site and the potential conditions at a restoration site:

- Functional similarity
- Climatological and hydrologic similarity
- Similarity in influences of human access, habitation, and economic activities and in the quantity and quality of water runoff from these activities to the wetland
- Similarity in the history of and potential for such activities as grazing, mowing, and burning
- Similarity in size, morphology, water depth, wetland zones and their proportions, and general vegetation types
- Similarity in soils and nonsoil substrates
- Similarity in access by fish and wildlife

A coast-wide reference monitoring system implemented to evaluate wetland restoration trajectories in Louisiana addresses the problem of identifying paired reference and restoration areas by providing an array of reference sites (Steyer et al., 2003; Cretini et al., 2012).

Enhancing predictive capability

Bradshaw (1987) proposed that the ability to predictably and successfully restore an ecosystem is the ultimate test – the "acid test" – of ecological understanding. The theory and practice of restoration is currently constrained by a lack of robust predictive capability (e.g., Cairns, 1995; NRC, 1992). Large-scale estuarine habitat restoration projects often require large financial investments and have low assurances of successfully meeting their goals. Research to increase predictive capability is occurring and has improved the predictions related to some selected habitat types (e.g., restoration of *Spartina alterniflora* marshes). The field of restoration ecology uses ecological experiments designed to predict the effect of a particular restoration effort (Jordan et al., 1987). This synthetic approach to restoration has found that the most powerful way of studying a habitat is to attempt to restore, repair, and adjust it so that it works properly. The very act of restoration provides new insight into the functioning of ecosystems. In turn, greater understanding of how ecosystems function enhances our ability to predictably restore them. The ability to predict what will be achieved by restoration, and when, requires long-term (10 years or longer) comparisons of restored and natural ecosystems. Such long-term research goes beyond systematic sampling by following promising leads and discarding hypotheses that do not stand up to testing (NRC, 1992).

Most projects attempt to set up *initial conditions* at a site that will likely lead to natural development of the site into a desirable state. Thus, a restoration action is the catalyst that sets natural succession in motion. Recognizing the dynamic nature of estuaries, the goal of estuarine restoration is to assist the "self-healing" capacity of the estuary, rather than to achieve some endpoint absolutely.

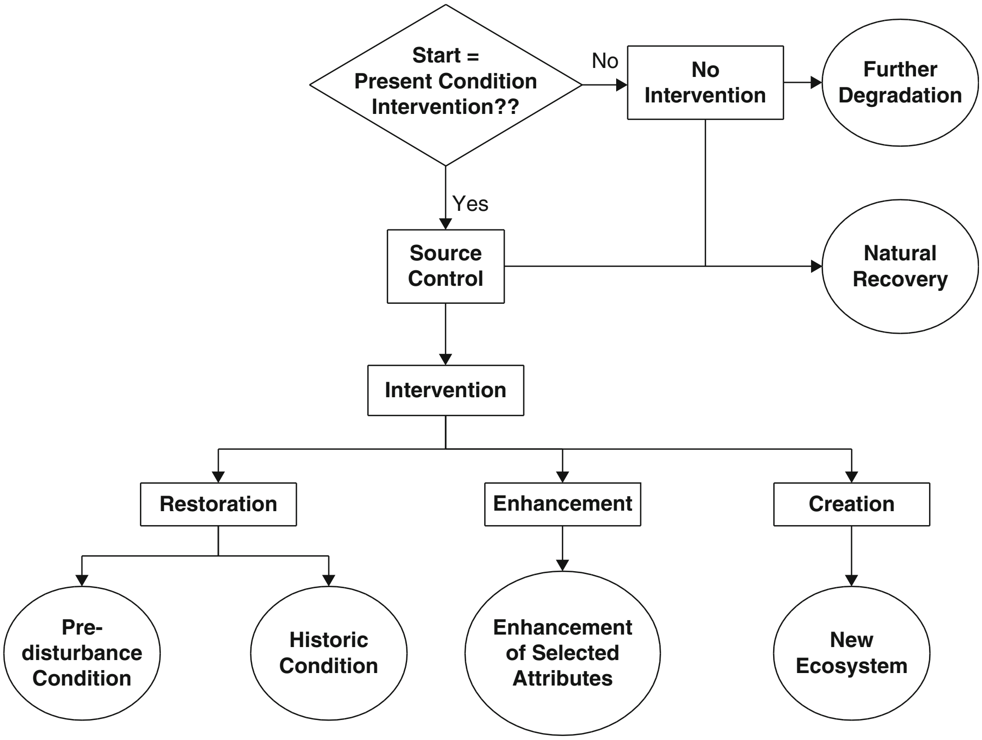
Landscape considerations in estuarine habitat restoration

To restore estuarine habitats, the ecosystem needs to be restored (NRC, 1992). Thus, for a habitat to be restored, the processes that contribute to forming and maintaining the habitat have to be active and intact in the landscape within which the restoration site occurs. For example, sediment from the watershed must be delivered to a newly connected former wetland site that has subsided in elevation while behind a levee.

Using the above example for seagrasses, to enhance seagrass distribution and abundance, it is necessary to reduce the input of inorganic nutrients from the watershed and from wastewater outfalls. Hence, strategies in the Chesapeake Bay have involved nutrient discharge abatement through enhanced treatment of wastewater and creation and protection of wetlands in the watershed (http://www.chesapeakebay.net/). These wetlands removed inorganic nitrogen that contributed to eutrophication of the estuary, which, through increased turbidity, caused loss of seagrass. Further, dense reefs of oysters that had been substantially reduced by overharvesting once were responsible for reducing turbidity in the Chesapeake Bay through their filter feeding. Currently, oyster reef restoration is part of the suite of actions taken to reduce turbidity and enhance seagrass recovery.

A principal landscape ecology concept is that most elements (habitats) within a landscape (e.g., watershed) function best when integrated with all other elements of the landscape. For example, habitat patch size, shape, location, and accessibility are critical to the formation of structure and realization of ecological functions. Human-caused fragmentation of natural landscapes into fewer and fewer smaller pieces leads to habitat shrinkage and less exchange of materials and species among habitats. Increasing fragmentation and decreasing habitat size can result in local extinction of some populations.

A landscape is a heterogeneous matrix of smaller habitats, and the arrangement, size, productivity, and resilience of these habitats within the matrix affect the flow of energy, animals, and materials through the landscape. Loss or degradation of one or more elements may lead to dysfunctional performance of the remaining elements. In deciding on restoration strategies and sites, it is useful to identify and consider the dysfunctional or absent elements.



Estuarine Habitat Restoration, Figure 3 Restoration strategies depending on the level of intervention. (From Thom et al., 2005).

The NRC (1992, pp. 347–348) concluded that "Wherever possible. . .restoration of aquatic resources. . .should not be made on a small-scale, short-term, site-by-site basis, but should instead be made to promote the long-term sustainability of all aquatic resources in the landscape." Successful restoration of self-maintaining habitat is more likely when the restoration effort is planned in the context of the landscape (Figure. 3). However, the complexity of recreating landscapes should not be used as an excuse for postponing restoration efforts.

Landscape ecology concepts applicable to estuarine habitat restoration include minimum area, shape, and corridors:

- *Minimum area* refers to the minimum area or size of a project required for an estuarine habitat to become fully functional (e.g., area needed to attract the species of interest, support the size of the species, their behavior within the habitat, required buffers, and stability over time).
- *Shape* refers to the shape of a patch or contiguous habitat that affects the types and number of species in the patch. Species can show preferences for edges or interiors of patches. As a patch increases in area, it usually develops a distinct interior and edge.
- *Corridors* are narrow strips of habitat that differ from the habitats on either side. Corridors can form very important protected routes of ingress and egress to habitats for species. They may also function as habitat for some species or as filters of disturbances (e.g., riparian buffer zones).

Actions to restore estuarine habitats

Actions to restore estuarine habitats have been categorized by Borde et al. (in preparation) and are quoted below. Although these actions are targeted for tidal marshes, they generally apply to other vegetated habitat types.

Hydrologic restoration

Natural hydrology is necessary for restoring functional coastal marshes, and this is often accomplished by returning tidal inundation via breach or removal of barriers such as dikes and levees or excavation of fill.

Removal of anthropogenic structures that hinder the hydrodynamic and geomorphic processes in tidal marshes has been applied as a management practice throughout the United States for decades. Traditional forms of this type of tidal marsh restoration include the removal or replacement of undersized or failing culverts as well as dike breaching activities. Channels with culverts bisecting tidal marsh habitats often create choke points for the flux of water, sediment, and nutrients. Furthermore, biological constituents associated with tidal marsh habitats can adversely respond to these constrained conditions. In a New England tidal marsh, vegetation and nekton exhibited a significant favorable response to the installation of larger culverts within the first year following construction (Roman et al., 2002).

Tide gate removal or modification
Tide gates are designed to constrain the natural hydrologic regime in tidally influenced habitats. Culverts are fitted with doors or flaps to allow freshwater to drain from upland sources while minimizing and often eliminating the inflow of salt water from adjacent tidal waters (Giannico and Souder, 2004; Giannico and Souder, 2005). Tide gates can allow hydrologic connection and permit limited fish access in projects where full connectivity is not possible.

Tidal channel development
Tidal restoration designs often include the plans for tidal channel development. This development can occur by means of passive formation, active creation, or a combination of both. Channels are passively or voluntarily created following a particular action that restores hydraulic and sediment processes to a tidal marsh. Hydraulic geometry and other indices provide useful guidelines for physical restoration and creation of estuarine tidal channels but do not clarify the ecological consequences of channel form (Hood, 2002; Diefenderfer et al., 2008).

Ditch plugging and filling
In U.S. east coast tidal marshes, ditching was historically implemented as a means for mosquito population control. The channelization and drainage of surface water within these ecosystems diminished natural water tables within marshes and ultimately led to the loss of salt marsh pond habitats (Adamowicz and Roman, 2005). Plugging ditches can create salt marsh ponds, whereas filling ditches can restore more natural tidal hydrology. Ecological effects of both of these methods should be given careful site-specific consideration (Corman and Roman, 2011).

Elevation manipulation
In a study by Cornu and Sadro (2002), the marsh surface of a diked and subsided estuarine wetland in Coos Bay, Oregon, was manipulated to examine structural and functional recovery at three intertidal elevations. Applying dredge material as a means to restore marsh habitats offers managers an opportunity to reclaim otherwise unusable material. Conversely, removal of fill accompanied by planting marsh species is an alternative method for elevation manipulation to restore tidal inundation and marsh vegetation (USACE, 2007b).

Plant propagation and reintroduction
Goals for restoration often include increases in native vegetation, habitat, aesthetics, and associated function. To that end, a common area of experimental research has focused on increasing the establishment, growth, and functional benefits of native tidal marsh vegetation. Soil amendments, including organic matter (e.g., composted kelp and municipal sewage sludge) or inorganic fertilizers (e.g., urea or ammonium nitrate), are experimental treatments that have been used in restoration sites with coarse soils, such as dredge material or sandy upland areas, where nitrogen is limiting (Callaway, 2001). The use of seedlings continues to be an effective approach for establishing diverse tidal marsh vegetation in smaller restoration projects.

Planting marsh vegetation offers managers an alternative to natural recolonization; however, this action may be cost-prohibitive for some restoration projects. Recovery and overall project success is largely dependent on site conditions such as marsh elevation, hydrodynamics, and the presence of nearby source vegetation for natural recolonization (Weinstein et al., 2001). Vegetative recovery can be influenced by these criteria, yet it is necessary for planners and resource managers to understand that the success of natural recolonization varies at multiple spatial and temporal scales.

Invasive species control
Nonnative plants in tidal wetlands threaten the natural biodiversity of coastal ecosystems. Purple loosestrife (*Lythrum salicaria*) and reed canary grass (*Phalaris arundinacea L.*) have a broad geographic footprint, plaguing both east and west coast tidal marsh habitats (Lyons, 1998; Blossey, 2002).

Fundamental components of habitat restoration projects

Fundamental components of successful habitat restoration projects and programs can be divided into five phases (Thom et al., 2011a): planning, implementation, monitoring, data management and dissemination of results, and adaptive management and project evaluation.

Planning

Planning includes the establishment of goals, objectives, and performance criteria for the project. Performance criteria often include time scale, spatial scale, structural conditions, functional conditions, self-maintenance, and the potential resilience of the system to withstand and recover from disturbance. The type of system to be restored is determined, and the site is selected. This involves examination of the historical or pre-disturbance conditions, degree of present alteration, present ecological conditions, and other factors. Conceptual ecosystem models are often used to help this process. The actual strategy used for restoration depends on the goals and objectives, as well as the condition of the landscape and the site. The level of physical effort, cost, schedule, contingency plan (i.e., in case something goes wrong), and engineering design are all part of project planning.

Implementation

Implementation means construction and begins with any required assessments, such as evaluating any onsite contamination. Construction is monitored by someone who is aware of the project goals to ensure that decisions made during construction result in the improvement of the system toward the goals.

Monitoring

Monitoring provides direct feedback about the development of the restored system with respect to established performance criteria. Appropriate field sampling methods are selected as needed for each performance criterion and metric. The selection of appropriate reference or control sites in the vicinity of the restoration project is also critical to the analysis of the monitoring data to identify trends that are not project-related. The NRC (1992, 2001) recommended that, when assessing change in a restored system over time, restoration monitoring programs should use science-based procedures and apply the following guidelines:

- Link assessment criteria to the goals and objectives of the project.
- Assess important wetland processes and functions or scientifically established structural surrogates.
- Base criteria on known conditions of the target or reference ecosystem.
- Establish assessment criteria before monitoring takes place and provide an indication of the expected degree of similarity between restored and reference sites.
- Incorporate the effects of position in the landscape.
- Choose criteria that are sensitive to temporal variation and spatial heterogeneity.
- Compare assessment results to reference sites and long-term data sets.
- Generate parametric and dimensioned units, rather than nonparametric rankings.
- Determine the monitoring period for reaching performance criteria a priori.
- Seek peer review for assessment criteria and methods.

The most specific guidance in the United States on the selection of restored wetland monitoring parameters comes from the NRC (1992, 2001), US Environmental Protection Agency (Kusler and Kentula, 1990; Kentula et al., 1992), Thom and Wellman (1996), Thayer et al. (2003, 2005), and Borde et al. (in preparation). The NRC (1992) recommended that for aquatic systems, at least three parameters be selected representing physical, hydrologic, and ecological features; too few parameters may provide insufficient information to evaluate performance or information that is difficult to interpret.

There are three basic questions to ask when selecting methods for monitoring: (1) does the method efficiently provide accurate data on physical and biological parameters, (2) is the method repeatable, and (3) is the method feasible within time and cost constraints? Any method used should have a documented protocol. The timing, frequency, and duration of monitoring are dependent on system type, complexity, and uncertainty. The monitoring program should be carried out according to a schedule that includes the program start and end dates, the time of the year during which field studies take place, and the frequency of field studies. Duration is dictated by the time the restoration project will take to develop to a point where it is clear that the project is meeting or will meet its goals.

Data management and dissemination of results

It is important to disseminate complete information about the project as widely as possible (Hackney, 2000). Aspects of the project are documented, and accurate and consistent record keeping helps document the effects of decisions and shows progress toward goals. Documentation benefits future projects by providing information that can minimize cost and maximize the probability of success.

Adaptive management and project evaluation

Monitoring documents project progress and identifies any problems that may affect achievement of the project goals. If the monitoring program identifies any deviation from the predicted trajectory of ecosystem development, adjustments can and should be made. Three general options are available to the project manager: *no action*, *maintenance or modification of the system*, and *modification of the project goals*. Adaptive management has been recommended at a national level and is used on many major restoration projects (e.g., Department of Interior, Williams et al., 2009).

Site stewardship and maintenance

It has been well documented that long-term stewardship and support are required to successfully restore estuarine habitats. Restoration of estuarine habitats requires time and vigilance to allow natural processes to reestablish themselves, and the stewardship must balance ecological and human needs (Borde et al., in preparation). As pressures on coastal systems increase, the need to maintain this balance will also become of greater importance.

Models used in habitat restoration planning

Conceptual models, geographic information system (GIS)-based models, operational numerical models, and population models are used to help plan and implement successful habitat restoration projects and programs.

Conceptual models

Conceptual models are used to develop performance criteria from goals and objectives and understand principal factors that control the development and maintenance of the habitat structure, the important habitat characteristics, and the functions for which the habitat is restored. Conceptual models help to forecast the effects of restoration actions compared to expected changes if no action is taken. The Chesapeake Bay Program restoration plan for submerged aquatic vegetation provides an example of how to relate performance criteria to goals using a conceptual model (Batiuk et al., 1992; Batiuk et al., 2000).

Geographic information system models

GIS-based models use knowledge of the spatial variability of factors that drive or limit species distribution, habitat quality, or ecosystem services. GIS-based models and applications are generally used during the project planning phase to assist with site selection. However, approaches, models, and applications vary widely. Process-based GIS models, such as the Wave Energy Model (WEMo; Malhotra and Fonseca, 2007), calculate quantitative physical parameters so that local data can be used to identify zones that meet thresholds or ranges of suitable values. Optimization routines, such as those within Marxan, examine different clusters of potential conservation areas to meet targets and minimize costs (Airame et al., 2003; Ball et al., 2009).

Models based on expert knowledge and ranked quantitative assessments of environmental stressors and functions are used to prioritize restoration areas (Diefenderfer et al., 2009). Participatory GIS approaches (e.g., NOAA's Habitat Priority Planner) use stakeholder criteria to visualize alternative scenarios (Bamford et al., 2009).

Recently, GIS models have been integrated with other software to expand visualization and analytical capabilities. For example, the Marine Geospatial Ecology Tools (MGET, Roberts et al., 2010) links ArcGIS with the statistical software R thereby enabling planners to predict spatial occurrences of sites and environmental conditions. The Gulf of Mexico Regional Collaborative integrates web models with conceptual model creation software (www.gomrc.org). With Envision, plug-ins enable development of future land-use scenarios that feed into habitat assessment models (Hulse et al., 2008). Finally, NatureServe's Ecosystem-Based Management Tools Network provides one of the most comprehensive interfaces to learn about and access spatially aware tool sets (www.natureserve.org).

Operational numerical models

Numerical models can help the planning process by facilitating sensitivity analysis and prediction of controlling factor conditions such as hydroperiod (e.g., Burdick, 2000; Yang et al., 2010). Hydrology is of critical importance to water resource projects and the science is well developed. Hydrologic modeling is frequently conducted during restoration project planning such as in the restoration of the Florida Everglades (Fitz et al., 1996). Numerical ecological models are less frequently used because the relationships among ecological parameters and the physical – chemical environment often are not well understood. However, ecological models have provided tools to describe predicted trajectories of ecosystem development under variable conditions. Modeling can complement field studies by improving the understanding of the relative effects of processes operating at different scales and thus improve project design, implementation, and adaptive management (Twilley et al., 1998). Models can also be used to help select performance criteria.

Population models

Population models of focal plant or animal species are used to help decide which processes are critical to explaining population and community dynamics, what form those processes take, and the value of parameters such as reproductive and survival rates. The process of developing a numerical model serves to formalize the current state of knowledge about the system along with critical uncertainties. Once developed, the model can predict the outcomes of management actions and support decisions. Finally, numerical models provide a mechanism for prioritizing research according to what will most improve the understanding of the system and the strength of the decision-making process.

Prioritization of restoration projects

Programs to restore habitats and ecosystems often have to prioritize which projects to undertake, what level of effort to devote to the projects, and where they should be located. Factors often included in systematic prioritization efforts include rare habitats that were once abundant, habitats that are critically important to and that allow direct access by endangered or threatened species, habitats that provide functions critical to an ecosystem (e.g., water-quality improvement), and habitats that would buffer disturbances from storm damage, flooding, etc. Ideally, projects are located at or near where these priority habitats once existed. Projects can be evaluated for their net improvement of the ecosystem. For example, projects to restore floodplain and surge plain habitats in the Columbia River estuary are prioritized by the habitat accessibility to juvenile salmon, capacity to support these fish onsite and offsite, and probability of restoration success (Thom et al., 2011b). The latter factor relates to the ability of the habitat to be resilient to withstand and recover from normal levels of natural disturbances (e.g., pulsed flood events). Cost can enter into these prioritization efforts, where the largest increase in function over the largest area would be realized for the lowest cost.

Costs of habitat restoration

Habitat restoration involves far more than planting vegetation. As noted above, it involves assessing, acquiring, and preparing the restoration site, addressing related safety considerations and permitting needs, and managing the project from start to finish. If a marsh does not exist at a site, there must be a reason why (Borde et al. in preparation). Often, restoring a marsh requires physical manipulations of a site. Heavy equipment may be necessary to remove obstructions to hydrology, reshape the ground, establish access points, etc. These types of activities can be extremely costly, requiring extensive engineering and construction expertise. In many cases, to save cost and enhance community support and involvement, volunteers are effectively used based on availability, interest, and training. In general, the size and complexity of a project adds cost to a project. Further, when extensive engineering is required to prepare a site as well as maintain a site over the long term, costs can escalate (Thom and Wellman, 1996). It is our experience that when costs for planning, land acquisition, implementation, maintenance, management, and monitoring are considered, habitat restoration project costs can range widely (i.e., $10,000–$500,000 per ha).

Long-term habitat viability

In general, the goal of a habitat restoration effort is to create a system that is relatively stable, persistent, resistant, and resilient. However, restoration of estuarine habitats must be viewed within the context of landscapes changing naturally with time. The long-term viability of a restored system is dependent upon a variety of factors inherent to the system, including stability, persistence, resistance, and resilience. Stability refers to systems that essentially show no long-term variability (i.e., tendency to move to another type of system). Persistence means the enduring nature (or presence) of a certain characteristic of the habitat relative to the passage of time. Finally, resistance and resilience, respectively, refer to the ability of the system to withstand and recover from disturbances. Owing to the buffering capacity associated with size of a site and the development of stabilizing features such as below-ground biomass of plants, larger, well-established habitats tend to be more stable, persistent, resistant, and resilient.

Summary

Because of the substantial losses and growing realization of the importance of estuarine habitats to endangered and threatened species as well as ecosystem services for humans, restoration of these habitats has been actively pursued for about the past three decades in the United States and other countries. The science of restoration has shown that restoring habitats requires the ecosystem (i.e., landscape) within which the habitats occur to be in relatively good condition. Restoration actions can range from simple restoration of hydrology to very active, complex, and expensive activities, including removal and reworking of sediments and elevations, removal of invasive species, and remediation of contaminants. Habitats can take from a few years to centuries to fully develop depending on the type of habitat and the conditions in the landscape. Restoration can be implemented by following a set of systematic steps. Stewardship is generally required to ensure long-term restoration projects success.

Primary information sources

To summarize this topic, we drew heavily from previously published documents, including chapters in a book edited by Kusler and Kentula (1990); a National Research Council report (NRC, 1992); chapters in a book edited by Thayer (1992), Shreffler and Thom (1993), and Fonseca et al. (1998); papers associated with a conference (Hackney, 2000), chapters in a book edited by Zedler (2001), the National Coastal Ecosystem Restoration Manual (Oregon Sea Grant 2002), Restore America's Estuaries (RAE) and NOAA (2002), Johnson et al. (2003), Thayer et al. (2003), Millennium Ecosystem Assessment (2005), and Thom et al. (2011a); papers associated with a conference edited by McGraw and Thom (2011) and Boesch (2006); and chapters in a book edited by Batzer and Baldwin (2012) and Borde et al. (in preparation). These documents present a synthesis of many published and grey literature.

Links to estuarine habitat restoration programs

North America

- Long Island Sound (http://longislandsoundstudy.net/research-monitoring/river-and-stream-bank-restoration-toolbox/restoration/)
- Chesapeake Bay (http://www.chesapeakebay.net/)
- Florida Everglades (http://www.evergladesplan.org/)
- Louisiana coast (www.coastalmasterplan.la.gov)
- San Francisco Bay-Delta (http://www2.epa.gov/sfbay-delta/bay-delta-action-plan)
- Columbia River Estuary (http://www.estuarypartnership.org/our-work/habitat-restoration)
- Puget Sound (http://www.psp.wa.gov/ and http://www.pugetsoundnearshore.org/)

Europe

- http://www.conference.ifas.ufl.edu/emecs9/Presentations/Monday/Salon%207-8/am/1055%20JP%20Ducrotoy.pdf

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Cross-references

Coastal Wetlands
Estuarine Deltaic Wetlands
Forested Wetland Habitat
Habitat Complexity
Habitat Loss
Saltmarshes
Tidal Freshwater Habitat
Wetlands

ESTUARINE SEDIMENT COMPOSITION

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Synonyms

Sedimentology

Definition

Sediment composition in estuaries relates to the different types of sediments found along and across an estuarine waterbody.

Introduction

Estuarine sediment composition is a complex topic requiring some background on estuarine classification. Estuarine classifications may be based on physiography (Kinsman and Pritchard, 1965), tidal range (Hayes, 1975), evolution (Dalrymple et al., 1992), morphology (Fairbridge, 1980), or morphogenesis (Perillo, 1995). For a classification to be of use, it must be based on physical and geological parameters that are common to all estuaries but that are different from one estuary type to another (Perillo, 1995). For example, tidal currents, winds, and sediments are all common enough to all estuaries, but variations of these parameters within a single estuary can be very large, so a classification based only on pattern distributions of these parameters would not be possible. Perillo (1995) argued that all previous estuarine classifications were too inclusive, so he introduced a genetic differentiation of estuaries which separated them into primary and secondary estuaries. Primary estuaries have not been changed significantly by marine processes, whereas secondary estuaries have evolved into different forms since their genesis. Estuaries can then be classified into a number of categories such as coastal plain estuaries, rias, fjords, delta front estuaries, structural estuaries, coastal lagoons, and others (see Perillo, 1995, for a more complete category list).

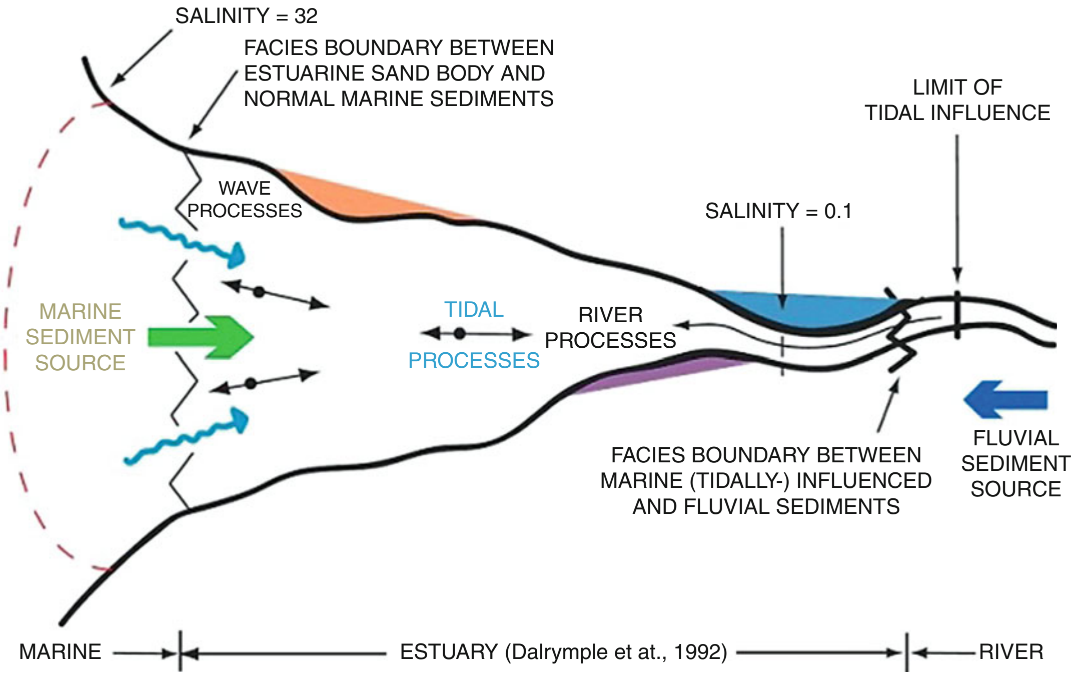
Sediment composition in estuaries is strongly influenced by the tidal range, wave heights (near the estuary mouth), sediment availability, and sediment transport processes, as shown in the Figure 1 (Dalrymple, 1992; Bianchi, 2013). Also, estuarine sediment composition is heavily dependent on the dominant source of sediment, which is either alluvial or marine. Independently of the estuary type, sediment composition in an estuarine environment varies axially and laterally, as well as vertically (Nichols and Biggs, 1985). Towards the head of the estuary, sediment composition includes silts, clays, plants, and roots, grading down to sand, gravel, and cobbles. Large clay and silt deposits may be found towards the mixed region of the estuary, together with sandy lenses and laminae. Nearer to the estuary mouth, the main sediments are marine sands with abundant cross-bedding, and tidally driven sandbanks may be found here. These sandbanks have a low-angle cross-bedding in fine sands with silt laminae. Lateral variations also occur, with shorelines composed of sand, gravel, shells, muds, plant fragments, and basal peat; subtidal flats composed of laminated muddy sands and sandy muds; and mid-channel environments dominated by coarse marine sands and massive cross-bedding. Sediment composition will now be discussed for some of the estuary types cited above.

Sediment composition in coastal plain estuaries

Most estuaries may be found in former river valleys on subtropical and temperate regions or former glacial valleys in polar and subpolar regions (Bokuniewicz, 1995). In pure coastal plain estuaries, sediment load provided by the rivers is small compared with the forces redistributing the material (e.g., current acceleration and gravity). Low-relief, coastal plain coasts favor the development of salt marshes and mangroves, with the subsequent infill of biological matter. Ocean waves will drive sand towards the mouth of the estuary, where complex systems of sandbars, spits, or barrier beaches may form. Marine sands may be transported landwards by overwashing of barriers by waves, transport due to tidal asymmetry, or transport by tides and estuarine circulation (Officer, 1981). In the central parts of the estuaries, fine-grained sediments may be found, consisting of submerged muds with abundant plant debris, or possibly fluid muds. The head of the estuary is characterized by fluvial sediment deposits with abundant plant debris and some brackish fauna. Some impression of the reversing tidal action will be preserved in the sedimentary structures (Goldring et al., 1978).

Sediment composition in delta front estuaries

Contrary to coastal plain estuaries, delta front estuaries have small forces redistributing the sediment compared to the amount of material available. This causes the formation of deltas, sediment protuberances of river-supplied sediment around the estuary mouth. A delta front estuary is the portion of the delta that is affected by tidal dynamics and water mixing (Hart, 1995). The factors controlling delta form and size include the characteristics of the source basin, which affect sediment size and supply, and the characteristics of the receiving basin, which affect the redistribution of that sediment. These factors are, in turn, controlled by climate, tectonics, and their interactions (Elliot, 1986). Attempts have been made to classify delta morphology in terms of fluvial, tidal, and wave influences (Galloway, 1975), but the variability found between deltas can only be explained by including other factors, such as the characteristics of the alluvial feeder system, the sediment grain size, the water depth of the receiving basin, and change in mean sea level (Postma, 1990; Orton and Reading, 1993). The alluvial feeder system supplies most of the water and sediment to the delta. Sediment yield will determine whether the delta will prograde, aggrade, or be transgressed. Sediment grain-size characteristics affect channel morphology, with the ratio of suspended load to bed load being a primary controlling factor (Schumm, 1977). Mud-dominated deltas form at the mouth of most long rivers; in humid tropical areas, alluvial systems supply large volumes of fine sediments to the delta; fan deltas, and braidplain deltas, are composed of coarse sediments (see Hart, 1995, and references therein). Finally, tectonic and climatic processes combine to redistribute the



Estuarine Sediment Composition, Figure 1 Estuaries facies model schematic (From Dalrymple et al., 1992; Bianchi, 2013 Nature Education).

sediments supplied by the alluvial systems. The Changjiang and the Mississippi are two very clear examples of delta front estuaries (Zhao et al., 2012).

Sediment composition in rias

There is some debate as to whether rias are estuaries or incised valleys, because only a small part of the ria is influenced by estuarine processes (Evans and Prego, 2003). However, the term ria is useful to define an incised valley, whether it is predominantly estuarine or marine. Rias may be found in rocky or cliffy shores that were not modified by alpine glaciation, but by subaerial erosion (Castaing and Guilcher, 1995). Apart from some exceptions, rias are found in the north of Spain, Brittany in France, South Devon and Cornwall in Great Britain, in southern Ireland, in Korea, in parts of China and Argentina, in the Red Sea, and the Mediterranean. The sediments in the rias in northern Spain and Galicia come from different sources. The sand is considered to derive from the erosion of rocks outcropping behind the coastal limestones and is maintained within the ria by the powerful Bay of Biscay surf. These sediments contain marine shells, which comprise 25–35 % of the sediments. Mud replaces sand in the inner reaches of the rias, with the usual presence of vegetated high marshes and bare low marshes. The rias in Brittany, France, may be categorized into four different sedimentological types: (1) pelitic rias (with sediments with median diameter of <50 μm); (2) sandy rias, or rias with large sand fractions coming from the sea; (3) widely open, or bay-like rias, with even larger sea influences; and (4) some microperiglacial, dwarflike rias (located in southwestern Brittany), with lengths of less than 2 km. In the southwest of England, the geological evolution is similar to the geological evolution of Brittany, and so Devon and Cornwall are girdled by a number of drowned valleys. Examples include the Exe, the Teign, the Dart, the Tamar and its tributaries, the Fal and the Carrick Roads, the Camel, and the Taw. The rias in Devon and Cornwall follow, as a whole, a pattern of sandy sediments near their mouths, and mudflats and high marshes in the inner reaches. Sand features are commonly found: Barnstaple Bay is fronted by the largest sand dunes in Devon and Cornwall, while the mouths of the Teign and the Exe are populated by complicated patterns of sandbars and spits (Steers, 1964). As for the Korean rias, the usual high marshes are rare or absent, due to land reclamations for rice cultivation. However, Korean rias present extensive areas of mud flats, sometimes several kilometers wide at low spring tide. Finally, little is known about the Argentinean rias, including the Deseado, San Julian, Santa Cruz, and Gallegos rias, all in Patagonia. The Deseado, Santa Cruz, and Gallegos have similar features. The last 18 km of the Deseado ria are covered by islands, tidal sandbanks, and small bays. Flood and ebb currents are turbulent enough to induce high turbidity of the water, loaded with volcanic clays. High tidal ranges cause large tidal flats, with outcrops and pebbly beaches along the rias margins. In contrast, the Santa Cruz ria is an ebb delta, with two ebb tidal channels.

Sediment composition in fjords

Fjords are often found in high latitudes and mostly on rocky shores. Fjords were created by the advance and retreat of glacial ice during the quaternary, and as such, they are immature systems that evolve and change over short timescales. In fjords, sediments originating from the continental shelf are less abundant that in other estuary types (Syvitski and MacDonald, 1982). This is because fjords typically contain one or more submarine sills that act as very effective barriers to marine sediments and because the compensation current is not near the bottom, as in other estuaries, but closer to the surface and thus does not lift sediments off the bottom and transport them up fjord. A substantial flux of organic matter in fjords may be linked to plankton that was transported into fjords by the compensation current, leading to plankton blooms (Petersen, 1978). According to Syvitski et al. (1987), fjords are incredibly efficient sediment traps, having retained around a quarter of the fluvial sediment delivered to the oceans over the past 100,000 years.

Sediment composition in coastal lagoons

Coastal lagoons, or bar-built estuaries, may be found in micro- and mesotidal coasts that are, or were in the near past, dominated by littoral processes. Sea-level variations, and the formation and growth of a lagoon barrier, play a significant role in the development of coastal lagoons. These features appear in coastal plains where a minor change in sea level causes major land inundations (Emery, 1967). Lagoons may be bordered by land, tidal flats, salt marshes, or mangroves, depending on the climate. Usually, the sediments in coastal lagoons tend to contain a large amount of organic matter. Seasonal, laminated muds dominate the bottom of template lagoons, except when the lagoon is shallow enough to be altered by wind waves. Sand content and wave action in coastal lagoons increase towards inlets, where ripples, mega ripples, dunes, and sand waves may form (Boothroyd, 1985). In addition, because of the high biological content, flocs play an important role in the sedimentary processes of coastal lagoons (Pejrup, 1988).

Summary

This contribution has focused on the sediment composition of estuaries. It has shown that all estuaries share some common features in relation to the sediment composition along and across the estuarine waterbody. However, this composition will depend on the geographical location of the estuary, on the estuary's origin, on the relative contribution of different physical processes and sediment sources, and on the landscape characteristics.

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Cross-references

Coastal Bays
Coastal Lagoons
Fjord
Ria
Sediment Transport
Sandbanks

ESTUARINE SEDIMENTATION

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Definition

An estuary was originally defined in geological terms as the seaward portion of a drowned valley system that receives sediments from both fluvial and marine sources and which contains facies influenced by tide, wave, and fluvial processes (Dalrymple et al., 1992). The estuary itself is regarded as the middle reach (or segment 2) of a complete incised-valley succession (Figure 1), being bounded seaward by the shoreline position at the beginning of the highstand and landward by the limit of marine influence (Dalrymple et al., 1994). A revised definition considers an estuary as a transgressive coastal environment at the mouth of a river, which is not necessarily linked to an incised valley. Instead, the two fundamental criteria for estuarine system generation are the existence of a relative sea-level rise leading to transgression and the occurrence of net landward sediment transport (Dalrymple, 2006).

Importance of estuaries

Modern drowned river valleys are generally populated environments that constitute the site of numerous human activities of economic importance. For example, they are emplacement sites of industrial practices (harbors, waste-disposal sites); they also constitute fragile environments of ecological significance, as they often support productive fisheries; and they also support recreational activities mainly related to tourism (Dalrymple et al., 1994). In addition, they can host significant quantities of hydrocarbon reserves produced by sand-dominated clastic reservoirs (Boyd et al., 2006). Finally, they can be used to predict future scenarios of environmental response to global change, as they are sensitive to sea-level and climate fluctuations (Tessier, 2012). In particular, they are good candidates to apply sequence stratigraphy concepts at the land-sea transition, allowing the precise identification of sequence boundaries (Zaitlin et al., 1994; Chaumillon et al., 2010). More recently, they have been investigated as paleoclimatic and paleoceanographic sedimentary archives (Troiani et al., 2011; Tessier, 2012).

Historical development

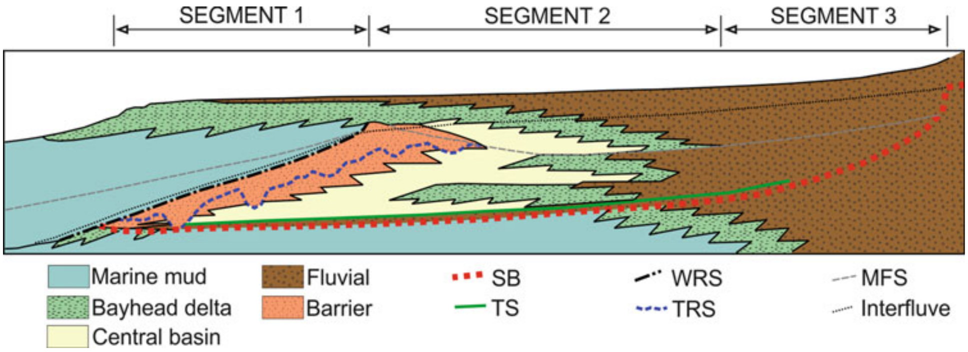
Historically, the study of estuarine deposits is logically linked to incised-valley systems. The early studies focused on the importance of erosional subaerial unconformities for the subdivision of the stratigraphic record (Dalrymple, 2006), with a fairly good documentation of the plan-view geometry of the valleys (Dalrymple et al., 1994). The initial research of estuarine deposits contained in the incised valleys highlighted the circulation patterns influenced by tides and the sediment dynamics, with distinctive upstream sediment transport (Boyd et al., 2006). However, the estuarine depositional systems were poorly documented (Dalrymple et al., 1994).

A significant step forward occurred between the 1960s and the 1980s, with the generation of models of estuarine sedimentation based on static facies models that emphasized the role of autocyclic processes (Figure 2), in particular in wave-dominated estuarine environments (Dalrymple et al., 1994; Boyd et al., 2006). The last mature phase of estuarine research was driven by the establishment of a conceptual facies model for estuarine systems (Dalrymple et al., 1992) and the subsequent development of sequence stratigraphy concepts, with a major emphasis on allocyclic factors leading to baselevel changes (studies included in the SEPM Special Publication No. 51). From these studies, a geological definition of an estuary was proposed, and a notable increase in the number of studies in estuarine settings was observed (Dalrymple et al., 1994).

Facies model

In estuarine environments, fluvial and marine processes interact, and as a consequence, a tripartite zonation is observed in most estuaries (Figure 2), reflecting specific energy levels and bedload transport patterns (Dalrymple et al., 1992). These include: (1) an outer zone dominated by marine processes; (2) a low-energy central basin; and (3) an inner, river-dominated zone. The central basin receives sediments both from the fluvial and marine systems and therefore is an area of net convergence.

There are two basic types of estuaries according to the dominance of marine processes: wave-dominated and tide-dominated systems (Figure 2). Wave-dominated estuaries exhibit the typical tripartite facies division, with an energy minimum in the central basin. The outer marine zone is occupied by a sandy body composed of several sedimentary systems, such as beach-shoreface,



Estuarine Sedimentation, Figure 1 Stratigraphic organization of a complete incised-valley succession, with subdivision in three segments. *Segment 1* represents the seaward portion; *segment 2* is the present-day estuarine system; and *segment 3* remains fluvial during the entire evolution of the system. Legend: *SB* sequence boundary, *TS* transgressive surface, *WRS* wave ravinement surface, *TRS* tidal ravinement surface, *MFS* maximum flooding surface. (Modified after Dalrymple et al. (1994)).

washovers, and tidal inlet and/or flood tidal deltas. The inner fluvial zone is generally covered by a bayhead delta. Tide-dominated estuaries show a slightly more complex pattern, as the energy minimum is less pronounced. The outer areas are dominated by elongate tidal sandbars, evolving landward to sand flats. The central part is characterized by a channel that shows a typical "straight-meandering-straight" pattern, where the meandering channel is equivalent to the low-energy central basin (Figure 2). Apart from the two basic types, there are other types of estuaries exhibiting strong physiographic control, such as estuaries generated by tectonic processes (also known as rias); rocky-coast estuaries, which show an irregular basement morphology (Chaumillon et al., 2010); and fjords, which are drowned glacial valleys (Reineck and Singh, 1980).

Stratigraphic organization

Sediments underlying present-day estuaries usually accumulate over an incised valley which is laterally related to a subaerial exposure erosional unconformity generated during the Last Glacial Maximum lowstand (Figure 1). This subaerial unconformity is considered the sequence boundary (Dalrymple et al., 1992). However, in some estuarine settings, such as the Rhine river mouth area, the sequence boundary has been linked to the previous sea-level lowstand during Marine Isotope Stage 4 (Hijma and Cohen, 2011).

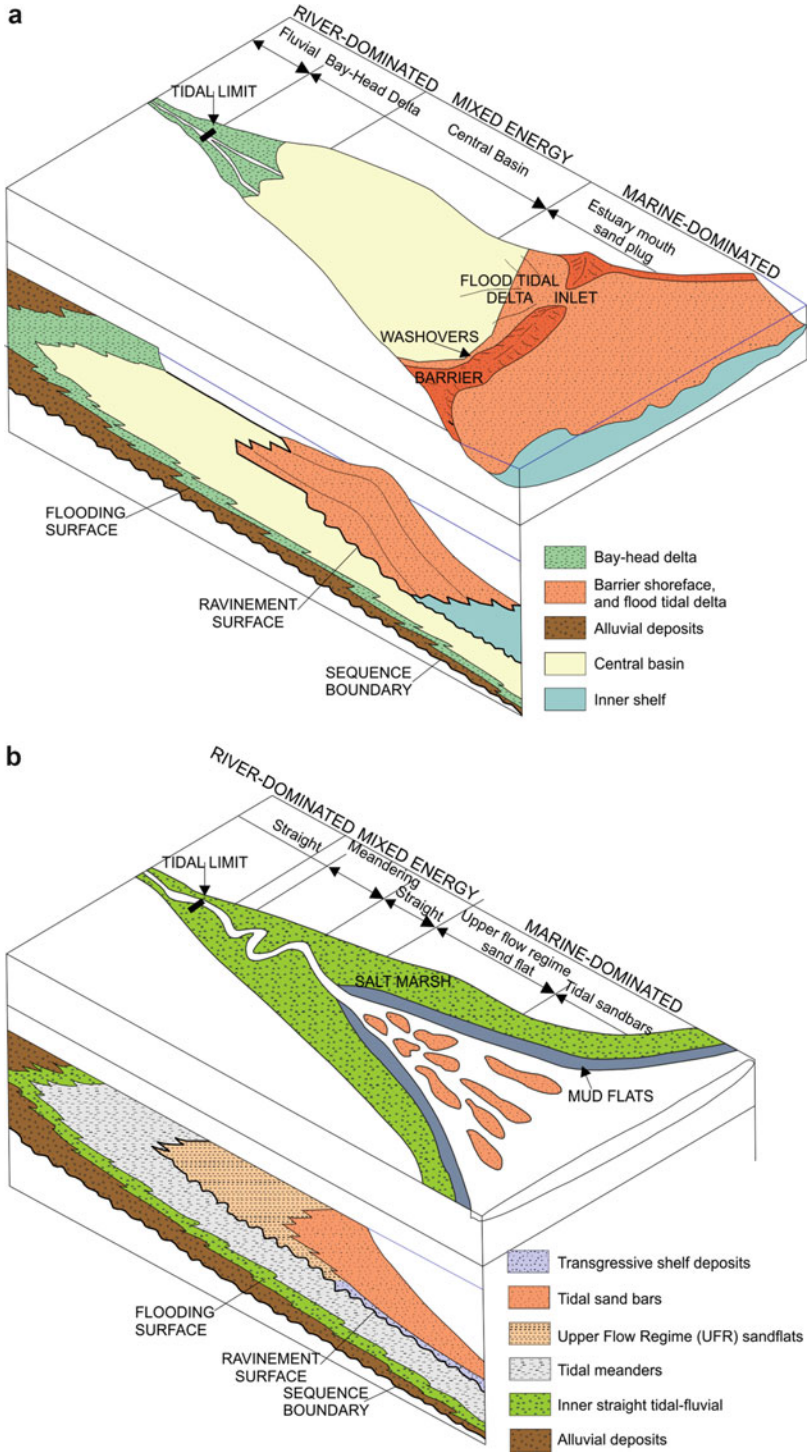
Although the original definition related the generation of the incised valley to the existence of relative sea-level fall, more recent approaches consider that: (1) some estuaries may not be necessarily formed as a result of river valley drowning and (2) the generation of the incised valleys may not be necessarily related to a relative sea-level fall, but to some type of physiographic (e.g., increase of river slopes) or climatic control inducing dramatic increases of sediment flux (Dalrymple, 2006).

Overlying sediments are composed by variable amounts of lowstand (LST), transgressive (TST), and highstand system tracts (HST), mainly depending on the interplay between the creation of accommodation space led by hydrodynamic factors, relative sea-level rise, and the fluvial supply (Dalrymple, 2006). However, in most estuarine systems the bulk of the sediment infilling is considered to be generated during the transgressive and early highstand phases (Dalrymple et al., 1992).

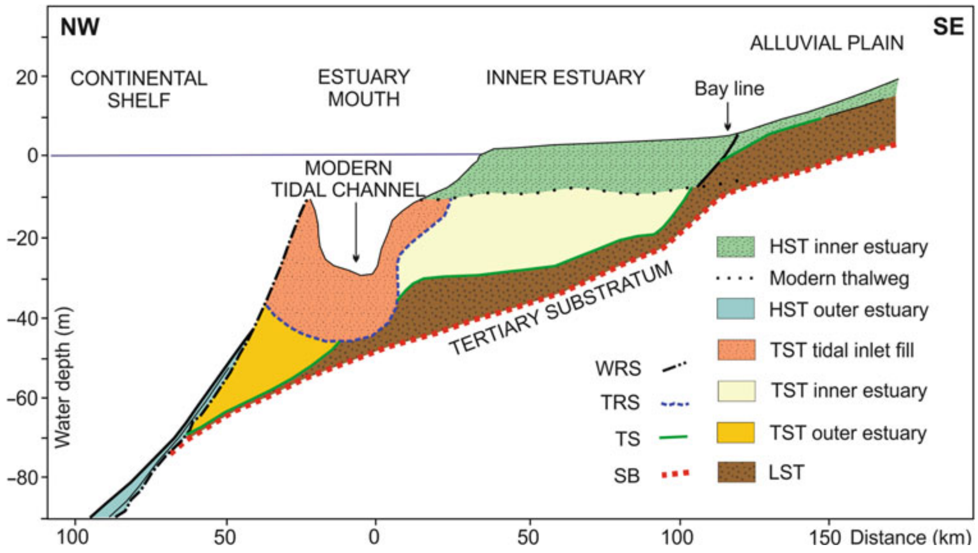
The lowstand unconformity may be covered by fluvial deposits that exhibit very significant variability, as they are missing in some paleovalleys, but constitute the bulk of the sediment infill in other cases (Dalrymple et al., 1994). Generally, thick LSTs are favored in narrow incised valleys (Vis and Kasse, 2009). Fluvial deposition may be restricted to lowstand conditions, when little fluvial sediment is deposited at the head of the estuary during transgression. However, most frequently, coarse fluvial deposits may be generated diachronously during lowstand to early transgressive conditions (Allen and Posamentier, 1993; Dalrymple and Zaitlin, 1994; Zaitlin et al., 1994).

The subsequent transgression would lead to estuarine sedimentation, the composition of which changes along the length of the estuary (Zaitlin et al., 1994). Estuarine sediments would be separated from the lower fluvial deposits (if present) by an initial flooding (or transgressive) surface (TS) (Figure 1). When fluvial sedimentation is restricted to lowstand conditions, the TS would coincide with the fluvial-estuarine transition (Lessa et al., 1998). In cases of intermittent transgression, several flooding surfaces will mark the ongoing valley drowning (Zaitlin et al., 1994). In cases when fluvial deposition continues during the transgression, the TS would lie within fluvial facies, and its identification would be difficult (Zhang and Li, 1996).

In general, higher amounts of TSTs are related to a combination of the following factors (Chaumillon et al., 2010): (1) existence of deep incised valleys, (2) large amounts of sediment supply, and (3) low tidal ravinement. The upper part of the estuarine sediments can be removed by transgressive ravinement, as a consequence of the generation of erosional surfaces, such as the tidal ravinement



Estuarine Sedimentation, Figure 2 Idealized facies models of estuarine systems, showing basic subdivisions between (**a**) wave-dominated and (**b**) tide-dominated estuaries. Wave-dominated estuaries exhibit the typical tripartite pattern, with a middle central basin separating areas dominated by marine (seaward) and fluvial (landward) processes. The distinction is less clear in tide-dominated estuaries, where the equivalent to the central basin is represented by a meandering channel in a zone with mixed energy. (Modified after Emery and Myers (1996)).



Estuarine Sedimentation, Figure 3 Stratigraphic architecture of the Gironde estuary, a mixed estuary which constitutes one of the classical examples of estuarine stratigraphy and shows common characteristics with wave-dominated estuaries. Legend: *WRS* wave ravinement surface, *TRS* tidal ravinement surface, *TS* transgressive surface, *SB* sequence boundary, *LST* lowstand systems tract, *TST* transgressive systems tract, *HST* highstand systems tract. (Modified after Lericolais et al. (2001)).

surface (TRS) at the mouth of the estuary that migrates landward or the bayhead diastem at the head of the estuary that migrates seaward (Dalrymple et al., 1994; Nichol et al., 1994). Additionally, a wave ravinement surface (WRS) may modify the top of estuarine deposits in distal settings, but this surface is generally out of the estuarine complex (Zaitlin et al., 1994).

Transgressive deposits may be finally buried by progradational estuarine deposits generated during the highstand (Figure 1), with an intervening maximum flooding surface (MFS) between transgressive and highstand deposits (Dalrymple et al., 1992). This change may be induced by a decrease in the rate of sea-level rise and/or by an increase of sediment supply (Emery and Myers, 1996), causing the MFS to be diachronous in most estuarine settings (Chaumillon et al., 2010). The expression of the MFS is variable according to the location along the estuary in both wave- and tide-dominated examples (Allen and Posamentier, 1993); most usually, its distinction is based on the recognition of a seaward shift of successive estuarine environments or more generically on the existence of a downlap pattern generated by progradation of overlying highstand sediments.

Simple versus compound estuarine infills

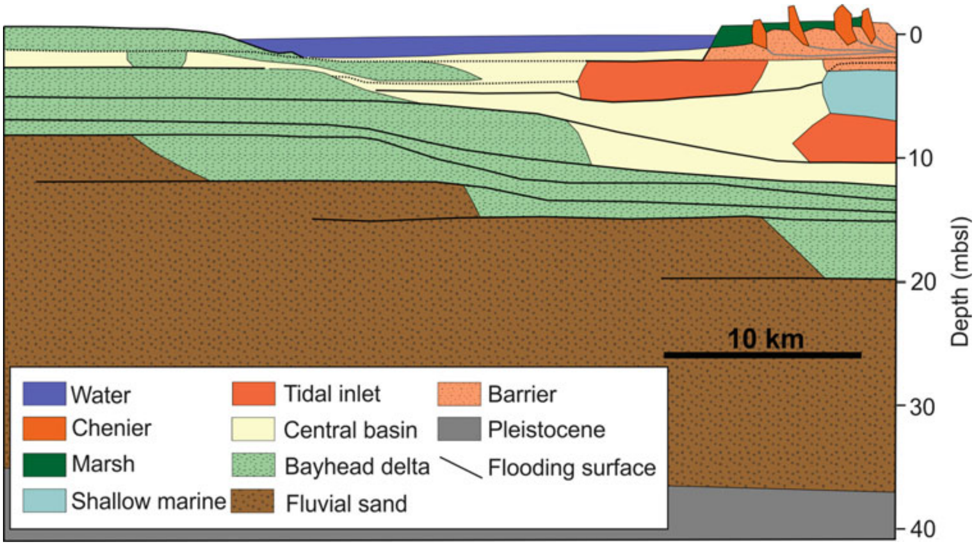
The above description of a typical infill of present-day estuaries refers to a simple situation, when the fill consists of a single depositional sequence. Alternatively, the fill can be compound, when it contains multiple sequences influenced by superimposed sea-level cycles (Dalrymple et al., 1994). There are several factors that influence the generation of simple versus compound infills such as rates of sediment supply, subsidence, and amount of transgressive/regressive erosion and depth of incision (Chaumillon et al., 2010). Thus, low values of those variables would tend to favor the generation of simple infills.

Compound estuarine infills preserve lower estuarine sequences below the most recent sequence boundary typically related to the LGM. These lower sequences tend to be composed of previous relative highstand deposits (e.g., genesis during Marine Isotope Stages (MISs) 3 and/or 5), such as barrier systems (Sloss et al., 2006). Alternatively, the development of lower sequences in estuarine settings may also be led by relative sea-level changes driven by isostatic rebound processes (Dalrymple and Zaitlin, 1994).

Wave-dominated estuaries

The development of the tripartite estuarine facies may occur both during transgressive and highstand stages, although the outer marine parts seem to be favored during transgressive conditions, whereas the inner fluvial parts seem to be favored during highstand conditions (Chaumillon et al., 2010). Mixed systems show both tidal and wave influences, but the resulting stratigraphic architecture seems to be similar to that of wave-dominated estuaries, due to the prevalence of the estuary-mouth sand body (Figure 3).

The stratigraphic architecture of wave-dominated and mixed estuaries was initially characterized by a prevalence of TSTs over HSTs (Allen and Posamentier, 1993). According to the model, the initial estuarine flooding caused the formation of bayhead delta facies at the base of the succession and at the head of the progradational estuary (Dalrymple et al., 1992). However, the stratigraphy of wave-dominated estuaries is primarily



Estuarine Sedimentation, Figure 4 Sediment infill of the Sabine-Neches Valley, considered as representative of the stratigraphic organization of most estuaries in the Gulf of Mexico. There, conditions of high sediment supply favor the development of bayhead deltas and central basin muds, separated by numerous flooding surfaces that document a complex transgressive history. The marine component is only developed during the last stages of estuarine infilling. (Modified after Milliken et al. (2008)).

characterized by the occurrence of the marine sand body (Figure 3), whose degree of preservation depends on the transgressive-progradational nature of the estuary (Dalrymple et al., 1992). A high sediment input and limited activity of the tidal inlet diastem would favor a significant generation and/or preservation of the marine sand body (Nichol et al., 1994), whose development takes place preferentially during the transgressive interval; thus, most of the estuarine mouth barrier is usually a component of the TST. The generation of a wave-dominated sediment body at the estuary mouth favors the accumulation of back-barrier muds (or central basin facies) or tidal-estuarine sands and muds in the case of mixed estuaries (Allen and Posamentier, 1993). These back-barrier deposits rest directly over the TS, which marks the transition to previous coarse-grained facies (bayhead deltas and/or fluvial facies), and evolve laterally to tidal flats and marshes.

The transgressive stratigraphic pattern records the change from central basin/estuarine sand and muds and lateral tidal flat/marsh deposits to fully marine conditions that are marked by the TRS (Allen and Posamentier, 1993) (Figure 3). This usually diachronous surface implies tidal channel formation and associated sediment removal that tends to be concentrated in the seaward portion of the estuary in the case of wave-dominated estuaries (Vis and Kasse, 2009), but may be more extensive in the case of mixed estuaries (Allen and Posamentier, 1993). In some cases, the marine sand and the TRS may extend significantly upstream in the estuarine environment, indicating hypersynchronous tidal characteristics (Lessa et al., 1998). The WRS has also been documented in wave-dominated or mixed systems, displaying a sharp contact of nearshore marine sands (Allen and Posamentier, 1993).

The development of a bayhead delta by the head of the progradational estuary (Dalrymple et al., 1992) tends to be fostered during highstand conditions (Allen and Posamentier, 1993), particularly under circumstances of enhanced fluvial supply and/or reduced wave activity (by sheltering). Bayhead deltas may coexist with distributary mouth bars, a seaward-migrating channel diastem and distal prodelta deposits that further contribute to the growth of the central basin facies (Nichol et al., 1994; Vis and Kasse, 2009). The development of central basin during highstand conditions implies that the MFS would presumably lie within these facies (Lessa et al., 1998). In mixed estuarine environments, tidal bars may also be developed in the middle estuarine sector during highstand conditions, prograding over previously deposited estuary-mouth sands (Allen and Posamentier, 1993; Tang et al., 2010).

The estuarine transgressive development may be different under conditions of high sediment supply, as recognized, for example, along the Gulf of Mexico coast and also in the Rhine Estuary (Hijma and Cohen, 2011). These estuarine infillings are characterized by deepening upward successions with fluvial deposits at the base overlain by estuarine deposits (Figure 4). However, the transgressive estuarine stratigraphy displays two main characteristics:

1. Numerous flooding surfaces separate a number of transgressive parasequences with a predominant backstepping stacking pattern (Figure 4). The origin of those flooding surfaces has been linked either to rapid glacio-eustatic

fluctuations, the most significant of them related to the 8.2 ka abrupt climatic event (Rodriguez et al., 2010), or to recent climatic changes such as transitions to more humid conditions (Simms et al., 2008) or to changes in wind strengths (Troiani et al., 2011).

2. The transgressive parasequences are mainly composed of bayhead deltas, evolving upward to central basin facies (Figure 4); the development of barrier systems has taken place only in the most recent estuarine filling stages and is generally restricted to the outermost parts of the estuaries (Anderson et al., 2008; Maddox et al., 2008; Milliken et al., 2008).

In contrast to the primary model that favored estuarine deposition during the transgression stage, a different scheme has been documented in numerous southeastern Australian wave-dominated estuarine systems, which is mainly applicable to the infilling of narrow incised valleys. In the case of low sediment supplies, these estuaries act as open coastal embayments during inundation of the incised valley. There, the main characteristic is the deposition of a basinwide transgressive sand sheet composed of washover and tidal channel sands and sand flats that extends much farther landward than the barrier of models described above (Sloss et al., 2005, 2006). The low fluvial supply does not favor bayhead delta construction during estuarine flooding (Heap and Nichol, 1997). The major development of estuarine infilling takes place during the highstand stage, due to continued growth of the sandy barrier, infilling of the tidal inlet, and development of a flood tidal delta. These all favor the development of estuarine infilling back-barrier facies. This process also triggers the progradation of bayhead deltas (with delta mouth bar sands and prodelta deposits) at the landward margins, induced by a relative sea-level fall (Sloss et al., 2005, 2006).

In the case of shallow incised valleys, but with conditions of high sedimentation rates, the main stratigraphic consequence would be the absence of fine-grained central basins due to enhanced fluvial sediment flux and/or low wave energy retarding the construction of subaerial barriers (Heap and Nichol, 1997). Instead, the occurrence of sediment bars would indicate localized tide dominance (Abrahim et al., 2008).

Tide-dominated estuaries

The stratigraphic patterns of tide-dominated estuaries are much less studied than the wave-dominated systems. The most significant examples of tide-dominated estuarine stratigraphy are provided by the Cobequid Bay-Salmon River Estuary (Dalrymple and Zaitlin, 1994) and several estuaries along the French coast of the English Channel and the northern Bay of Biscay (Tessier, 2012).

The sequence boundary of tide-dominated estuaries is a fluvial valley related to a sea-level fall, conforming to the generic definition (Figure 5). This is possibly due to the fact that narrow-valley features are necessary to cause an increase of tidal influence and by extension to generate a tide-dominated estuary. The LST deposited at the bottom of the fluvial valley is generally strongly reduced due to very strong tidal transgressive ravinement (Tessier, 2012).

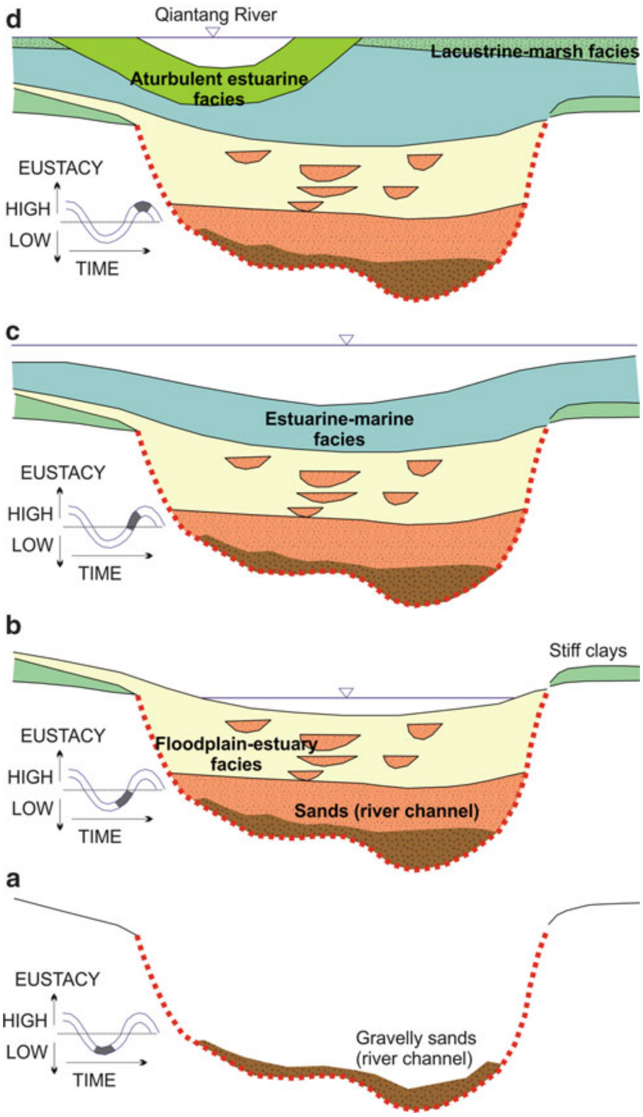
The main stratigraphic component of tide-dominated estuaries is the marine sand body composed by tidal sandbars (Figure 5). Landward, the tidal sandbars may evolve to point-bar deposits sandwiched by straight tidal-fluvial deposits (Dalrymple et al., 1992), but bayhead deltas are not present (Dalrymple and Zaitlin, 1994). According to local hydrodynamic and sediment supply conditions, the bulk of the infill may be related either to the TST or to the HST.

Stratigraphic patterns of tide-dominated estuaries derived soon after the development of the estuarine facies model, and stratigraphy depicted the main development of estuarine facies as preferentially taking place during the transgressive interval (Dalrymple and Zaitlin, 1994). The dominance of transgressive over highstand deposition was linked to the development of extensive sediment facies during the estuarine flooding (Figure 5), from significant fluvial transgressive deposition to tidal ridge systems related to the transport of sediments into the estuary due to tidal current amplification (Zhang and Li, 1996; Lin et al., 2005). In these cases, the estuarine HST is composed of the most recent (thin) infilling stages that tend to bury the transgressive infilling (Figure 5).

However, more recent case studies have shown that the bulk of the infill may have occurred under highstand conditions (Figure 6), and the TST is an aggradational unit of reduced volume. This pattern is usually related to strong tidal ravinement and deep occurrence of the TRS, which cause the erosion of the TST, and the development of thick highstand tidal sand bodies, resulting in the dominance of the HST, particularly at the estuarine mouth (Tessier, 2012; Tessier et al., 2012).

The TRS is the most important stratigraphic surface in the sediment record of tide-dominated estuaries, as it extends through the entire estuarine section (Figure 6), although the TRS may not incise as deeply as in wave-dominated or mixed estuaries due to the absence of channel constriction at the estuary mouth (Dalrymple and Zaitlin, 1994). Depending on the major development of estuarine facies, the TRS may be amalgamated with other estuarine surfaces. For example, in the case of major estuarine infill during transgression, the TRS is amalgamated with the TS, whereas during major estuarine development during the highstand, the TRS is amalgamated with the MFS (Figure 6). In contrast, the WRS tends to be absent or poorly developed, as a wave-dominated shoreface is not present (Dalrymple and Zaitlin, 1994).

Another significant stratigraphic feature of tide-dominated estuaries is the fact that most of them contain wave-dominated facies, which may exhibit temporal or spatial variability. For example, several tide-dominated estuaries have wave-dominated environments, such as coastal barriers or central muddy basins during the transgressive stage, indicating that most tide-dominated estuaries undergo a significant change during their development, most possibly due to tidal amplification



Estuarine Sedimentation, Figure 5 Characteristic evolutionary stages of a tide-dominated estuary, exemplified by the incised-valley succession of the Qiantang River with the following phases: (**a**) formation of the incised valley with moderate fluvial deposition during lowstand conditions; (**b**) generation of transgressive channel-infilling and floodplain-estuary sequences, due to retrogressive aggradation and with development of sand bodies interpreted as sand ridges due to enhanced tidal activity; (**c**) estuarine burial by estuarine-marine sediments during final transgression; (**d**) generation of estuarine sandbars during the final sea-level stabilization. (Modified after Lin et al. (2005)).

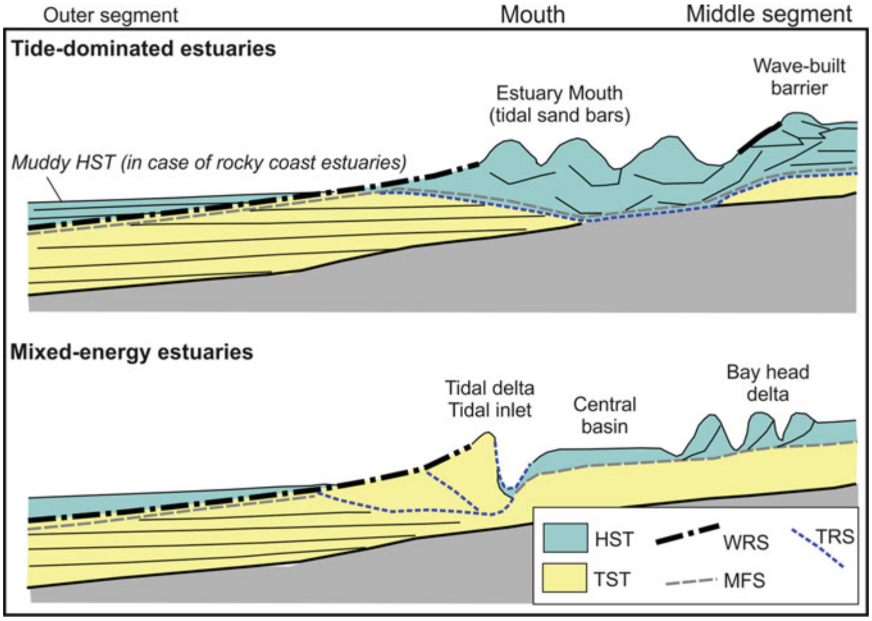
estuarine flooding. In other cases, tide-dominated facies generated during the highstand interval laterally evolve to wave-built coastal barriers (Tessier, 2012); this lateral change is related to the lower incision which leads to less active tidal dynamics, favoring wave activity (Tessier et al., 2012). In addition, tide-dominated estuaries generally show a dominance of sandy over muddy facies, due to tidal activity in the estuary (Chaumillon et al., 2010).

Rias and rocky coast estuaries

These types of estuaries display a dominant control of bedrock morphology during the entire infilling of the estuarine system (Chaumillon et al., 2010). Due to their deep valleys, rias are prone to record compound infills, containing sequences prior to the LGM and multiple postglacial transgressive deposits, as in the case of the Ría de Vigo in the Galizian Margin (García-García et al., 2005).

Fjords

Fjord sediment infills also have an overall transgressive-regressive architecture. However, as a major difference to lower-latitude estuaries, the infill of fjords is primarily affected by isostatic uplift processes, causing a relative



Estuarine Sedimentation, Figure 6 Stratigraphic patterns of estuaries in the Bay of Biscay, highlighting the distinction between tide-dominated estuaries characterized by main development during the Holocene highstand, in contrast to the architecture exhibited by mixed estuaries, where the development took place during the transgressive interval. Legend: *HST* highstand systems tract, *TST* transgressive systems tract, *WRS* wave ravinement surface, *TRS* tidal ravinement surface, *MFS* maximum flooding surface. (Modified after Chaumillon et al. (2010)).

sea-level fall at the end of the development, generating a forced-regression deposit. The other significant feature of many fjord infills is the frequent occurrence of gravity-controlled deposits, due to the influence of rapid isostatic rebound during and after the deglaciation (Hjelstuen et al., 2013). In general terms, three main depositional units are distinguished in fjord infills (Corner, 2006):

1. Deglacial transgressive systems tract, formed by retrogradational transgressive infilling during deglaciation. The infilling is primarily derived of glacial meltwater from the retreating glacier.
2. Deglacial highstand systems tract, formed by progradational infill at the head of the fjord during short-lived highstand conditions at the end of the glaciar retreat.
3. Postglacial forced regressive systems tract, formed by aggradational and progradational fluvio-deltaic infilling led by relative sea-level fall and accompanied by emergence.

Controlling factors

The simple facies and stratigraphic models described above cannot cover all the causal factors observed in nature. The factors that control the development of estuarine stratigraphic features, including different methods of formation and preservation of sediment bodies and stratigraphic surfaces, are the following:

1. Bedrock valley morphology. The shape of the valley may influence its subsequent transformation into an estuarine system and the different development of depositional systems (Dalrymple et al., 1992; Boyd et al., 2006). The existence of irregularities caused by sedimentary or tectonic processes may favor the development of wave-dominated estuaries (Dalrymple, 2006). In contrast, funnel-shaped valleys with a high length-width ratio tend to develop tide-dominated estuaries, as they favor the generation of hypersynchronous estuaries (Tessier, 2012). Additionally, the depth of the incision controls the preservation potential of infilling deposits, particularly of the lowermost such as LSTs and/or TSTs (Chaumillon et al., 2010).
2. Interrelation between hydrodynamic processes. Estuarine facies distribution and deposit architecture of the two end-member states (wave versus tide dominance) are variable according to the relative importance of waves and tidal and fluvial currents (Dalrymple et al., 1992). For example, tidal accommodation controlled by tidal range and depth of tidal ravinement surface is a major control on the preservation of estuarine deposits in tide and mixed settings (Chaumillon et al., 2010; Tessier, 2012). In wave-dominated estuaries, wave activity is a major factor that controls barrier growth (Chaumillon et al., 2010).
3. Sea-level fluctuations. The relative sea-level patterns govern the overall stratigraphic change observed in most estuaries from transgressive to regressive conditions, related to a significant decrease of relative sea-level rise. The inundation of the fluvial valley during sea-level rise may lead to a tidal resonance

process and to the transformation from a wave- to a tide-dominated estuary (Dalrymple and Zaitlin, 1994).

4. Sediment supply. In general terms, the amount of sediment supply will control the nature of sedimentation in estuarine environments. The most extreme case would be a give-up estuary, where conditions of very reduced sediment supply would be insufficient to generate an incised-valley fill (Cooper et al., 2012). Low-supplied systems are mainly filled with marine sediments, such as the case of French estuarine environments (Chaumillon et al., 2010). In general, those low-supplied estuaries show reduced and absent TSTs. The variability of sediment supply is particularly important during estuarine transgression, as high-sediment supply may account for significant development of estuarine mouth barriers under wave dominance, or high fluvial supply may favor anomalously thick TSTs. In contrast, highstand conditions are generally characterized by increases of sediment supply, as the decrease in the rate of sea-level rise favors the influence of fluvial sedimentation, accounting for the generation of bayhead deltas that may develop seaward into prodeltaic environments and ultimately leading to significant fine-grained sediment delivery to the shelf. In addition, carbonate production may also be enhanced by highstand conditions, thus increasing the volume of the HST (Tessier, 2012).
5. Climate change. The impact of recent climatic variability on estuarine sediment infillings has been addressed in several recent studies, mainly in northwestern Europe and the Gulf of Mexico estuaries. For example, periods of increased storminess at millennial timescales are mainly recorded in marginal estuarine highstand facies, where the destruction of coastal barriers adjacent to estuarine settings and tidal incisions has been documented (Tessier et al., 2012). The activity of intense storms may also be imprinted in the estuary mouth through the formation of wave-dominated sandy facies (Tessier, 2012). In the Gulf of Mexico, several recent transitions from dry to humid conditions appear to be responsible of sediment supply reductions to estuarine settings (Anderson et al., 2008; Simms et al., 2008).
6. Human influences. The exploitation of drainage basins for human activities in the last few centuries is argued to have caused significant modifications of estuarine sediment rates. In particular, a recent period (i.e., the last 1,000 years) of increased fine-grained deposition documented in several French estuaries has been related to increased soil erosion by deforestation and agricultural practices (Tessier, 2012). The intensification of agricultural practices as triggering mechanism for bayhead delta growth has also been documented in some estuaries along the Gulf of Mexico coast (Anderson et al., 2008).

Recent and future trends

Future research on sediment infill of present-day estuarine systems should be directed at improving the definition of estuarine systems by using different techniques and/or approaches such as the following (Boyd et al., 2006):

1. Numeric models that can provide a quantitative approach to the major operating processes (sediment flux versus relative sea-level changes) and can be used to predict the future estuarine behavior.
2. 3D seismic data and seabed imagery can enhance our visualization of the complexity of estuarine sedimentary environments.
3. Other geological approaches such as brackish ichnology and petrological and chemostratigraphical studies, among others, may be helpful for the recognition of estuarine facies and for the subdivision of the estuarine record into different sequences.
4. Improved knowledge of the longitudinal variability of the estuarine infill, as most of the present knowledge involves lateral variability (Tessier, 2012).
5. Better understanding of the influence of anthropic activities in the development of estuarine stratigraphy, which is thus far poorly documented (Tessier, 2012).

Summary

The sedimentary infill of present-day estuaries has been mostly generated during the course of the postglacial sea-level rise (simple infill), although in some cases older sequences may be preserved in deep incised valleys (compound infill). The base of the infill is generally represented by a Last Glacial Maximum incised valley, which may be covered by lowstand (to early transgressive) fluvial deposits. The record of subsequent transgressive to highstand sediments and surfaces will be mostly determined by the hydrodynamic conditions, with two end-member cases (wave- versus tide-dominated estuaries).

Most wave-dominated estuaries show a major development of transgressive deposits, either with preferential occurrence of the outer marine sand body, in cases of moderate tidal ravinement, or with major generation and/or preservation of bayhead deltas and central basin facies, under conditions of very high sediment supply. A different picture has been provided in narrow incised-valley estuaries, where the transgressive interval is recorded by a widespread sand sheet, and the major development of the estuarine stratigraphy takes place during highstand conditions.

In the case of tide-dominated estuaries, two distinct stratigraphies may be observed. A major development of transgressive deposits is linked to tidal activity (and widespread occurrence of the tidal ravinement surface) that favor the generation of tidal sand ridges. Alternatively, a very strong tidal ravinement would cause the erosion of most of the transgressive deposits and major deposition of estuarine facies (tidal sand ridges and fluvial point bars) during the subsequent highstand.

Other estuary systems show peculiar stratigraphic features. For example, rias and rocky-bound estuaries tend to preserve older sequences in the deep estuarine sections.

Fjords also show a transgressive-regressive sequence, but with final development of forced regressive deposits because of the influence of isostatic rebound processes.

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Cross-references

Estuarine Sediment Composition
Fjord
Sediment Budgets
Sediment Grain Size
Sediment Transport
Sedimentary Structures
Stratigraphy of Estuaries

ESTUARINE SUSTAINABILITY

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Definition

Estuary sustainability can be defined as the ability of an estuarine ecosystem to maintain ecological processes and functions, biological diversity, and productivity over time. However, this is the rather narrow ecological definition of the term, and there are multiple contemporary definitions that go beyond science because they involve economics and values. Estuary sustainability, therefore, must be examined, and measured, in the local and regional context but with an eye on the international aspect because clearly broad-scale effects at the ecological, economic, and value level can quickly spread from one part of the world to another.

Introduction

In the general context, sustainability often relates back to the often-interchangeable terms "sustainability" and "sustainable development" and can be traced to the United Nation's Brundtland Commission report of 1987, officially titled Report of the World Commission on Environment and Development: Our Common Future (United Nations, 1987). The Brundtland Commission report identified components of sustainable development: ecology, economy, and equity. A brief discussion of each follows, with specific reference to estuaries.

Ecological sustainability

Measurement of ecological sustainability is closely tied with the concept of estuary conservation (qv). The structure and function of estuary ecosystems are important attributes to consider, as well as maintenance of the capacity to adapt to change (resilience). Ecological processes, such as productivity in estuaries, are key to functions such as food production from their habitats. In addition, the maintenance of biological diversity as a structural component in estuaries has emerged as an important aspect of sustainability. Loss of biodiversity in the tropical mangrove ecosystem from tin mining, aquaculture, and forestry is an example of the problem in tropical estuaries (Macintosh et al., 2002). The Convention on Biological Diversity, developed by the United Nations Environment Programme, has been ratified by over 100 countries around the world to date (Convention on Biological Diversity, 2013) and is projected to improve estuary biodiversity management.

Economic sustainability

A challenging economic component of sustainable development relates to the term's frequently cited definition: "meeting the needs of today without compromising the needs of future generations." Most societies have a need for industrial activity or some form of economic activity to generate income. Measuring estuary sustainability in this economic context is a challenge as the standard macroeconomic metrics such as gross domestic product (GDP), which are typically expressed as monetary values, are difficult to apply to ecosystem services. A variety of ecosystem-based metrics such as critical natural capital (CNC) have been developed (Hak et al., 2007). Estuaries are often included as CNC in analyses of economic sustainability indices around the world (e.g., the USA, Liu et al., 2010). Because of the importance of ports and harbor development in estuaries around the world for coastal economies, there has been an emphasis on "greening" ports to reduce the impact of shipping on CNC (e.g., processes to minimize air pollution from running engines while in harbor) as well as development of indicators of harbor sustainability (e.g., Peris-Mora et al., 2005). The siting of new harbors to enable ecological and economic sustainability is an important aspect of integrated coastal zone management (qv).

Equity sustainability

The impartial or equitable allocation of renewable resources to various people in the estuary is a key concept of sustainability but is also very challenging to measure. As well, because estuary ecosystem functions depend on river conditions (maintenance of stream flows) as well as ocean factors (e.g., sea level rise), the scope of communities that are stakeholders in the estuary is very broad. The allocation of fisheries resources is perhaps where most of the emphasis and analyses of estuarine equity have been focused, especially where policies strive to reduce catch and limit access to the resource in efforts to maintain stocks. Limiting access to fisheries frequently has the greatest impact on the small-scale, traditional fisher (e.g., Cochrane, 2000). This is an example where equity needs are likely not being met. Ecotourism in estuaries is another business area that is developing inequitably according to some authors (Jamal et al., 2006), and they suggest the fostering of ecotourism's alternative potential for improving environmental, social, and cultural well-being of people living on estuaries.

Summary

Achieving sustainability in estuaries is a test to coastal societies around the world and requires constant attention using adaptive management to ensure the "three pillars" of the concept are achieved. A good strategy to move forward and tailor an individual estuary sustainability plan to local requirements and values is to compare the various institutional arrangements in place in different countries with varying socio-ecological systems as they to strive for estuarine sustainability (Ostrom, 2009).

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Cross-references

Estuary Conservation
Sustainable Use

ESTUARINE TOTAL ECOSYSTEM METABOLISM

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Definition

Estuarine total ecosystem metabolism is the sum of all metabolic processes associated with primary production, secondary production, and respiration in an estuary.

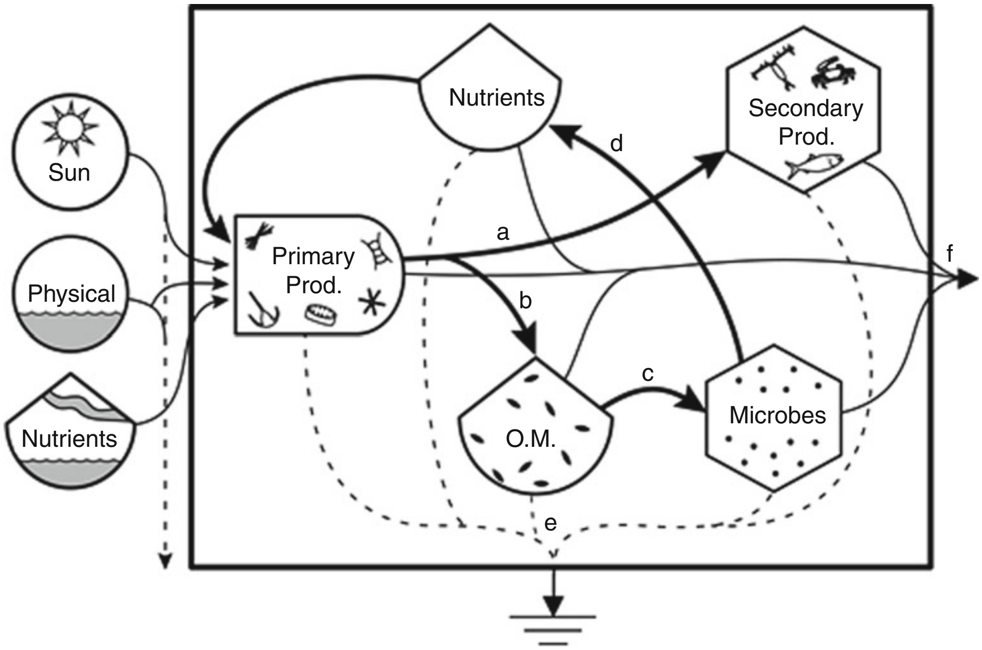
Introduction

Each day as the sun rises and retires the beautiful green bays like great creatures breathe in and out. By day photosynthetic production of food and oxygen by plants is plentiful, but day and night there is also a furious feasting (Odum and Hoskin 1958).

Every living system, from the scale of an individual organism to that of a complex and interconnected ecosystem, consumes energy as a means to create or process organic material. These transfers of energy and organic material production, consumption, and use are the functions of metabolism. The rate of metabolism for a given system can shed light on questions concerning its function and interaction with other systems. What is the maximum energy or product yield of an estuary? Will the rate of estuarine production respond to changes of external inputs such as sewage diversion or modified river flow? Here we briefly review the historical development of metabolism studies for estuaries and the range of techniques researchers have used as a means to answer questions on estuarine system function, connectivity, and change.

Ecosystem metabolism

Measurements of the components of ecosystem metabolism (e.g., production, respiration, heat loss), as well as the sum of the absolute values of these components (total ecosystem metabolism or TEM), were an integral part of the development of the ecosystem concept in ecology (Figure 1). As the idea developed that plants, animals, and their surrounding physical environment could be described and quantified as distinct units of energy or resources, the question naturally arose as to whether ecosystems were in equilibrium, where the inputs to the system equaled the outputs exported from the system (see Golley, 1993). Some of the earliest studies exploring this question were conducted by H.T. Odum in Florida springs, but also by H.T. and E.P. Odum at the Eniwetok Atoll, Marshall Islands (Odum and Odum, 1955). The transfer of energy within an ecosystem was expressed as the ratio of primary production (P) to respiration (R), where primary production is the organic matter made via photosynthesis using the energy of the sun, and respiration is the breakdown of organic matter which is then converted back



Estuarine Total Ecosystem Metabolism, Figure 1 An energy flow diagram of the components of ecosystem metabolism in an estuary. This diagram is a simplification for the purpose of illustrating estuarine total ecosystem metabolism (TEM), and thus only the major energy flow pathways are represented. We acknowledge that other pathways exist and that feedback loops among trophic components are common. The estuary is enclosed by the box with major energy sources to the estuary listed to the left of the box (sun, physical inputs, nutrients). Primary producers use the physical inputs to create organic matter (O.M.), which is then consumed by secondary producers and microbes. A portion of this O.M. is converted back to an inorganic state (nutrients). NEP is the rate of primary production minus the rate of respiration. Respiration is the sum of all activities that process organic material (a + b + c + d + e). Energy or carbon losses occur through exchange with the ocean and burial (f) and heat loss (e). TEM is the sum of all energy flow pathways within the estuary. Some symbols used in diagram courtesy of the Integration and Application Network, University of Maryland Center for Environmental Science.

into inorganic components (Figure 1). Whereas primary production can only be accomplished by photosynthetic organisms (e.g., plants), respiration is conducted by all active organisms in the community (plants, animals, and microbes). These variables (TEM, P, R) are typically expressed as rates such as a mass of carbon (or unit of energy) per unit area per unit time (e.g., mg C m^{-2} d^{-1}). An autotrophic ecosystem creates more organic matter than is broken down (production exceeds respiration, $P/R > 1$), whereas a heterotrophic ecosystem consumes more organic matter than is produced (i.e., organic matter inputs exceed production and thus $P/R < 1$). From a system perspective, autotrophic estuaries are net exporters of organic material, while heterotrophic systems are net importers. These concepts became particularly important with the advent of the environmental movement in the 1970s and 1980s, when research scientists began to explore the links between coastal ecosystems (estuaries, salt marshes, mangrove forests, etc.) and adjacent commercially important marine fisheries.

Ecosystem metabolism measurements

While total ecosystem metabolism is the sum of all of the energy transferred by creating, utilizing, and decomposing organic matter, measuring each of these components individually is not practical in estuarine systems due to vast ranges and fluctuations over temporal and spatial scales, as well as functional and structural differences within communities (e.g., seagrass vs. phytoplankton, menhaden vs. copepods). An indirect and integrative measurement is net ecosystem production (NEP), sometimes also referred to as net ecosystem metabolism (NEM), which is the rate of gross primary production minus the rate of respiration for all biologic components. To determine gross primary production (GPP), the amount of organic matter respired (R) is added to the NEP, where $GPP = NEP + R$. Essentially, GPP is an accounting of the daytime production measured in the water column (NEP) plus the production created but then respired before it could be measured (R).

A classic method in phytoplankton-dominated estuaries is the light and dark bottle technique, where transparent and opaque bottles are each filled with estuarine water and suspended in the water column for a specific period of time (usually hours). Oxygen concentrations within the bottles are measured at the beginning and end of this incubation, and the increase in oxygen in the transparent bottle (where both photosynthesis and community respiration have occurred) is the NEP, and the decline in oxygen in the dark bottle represents R. GPP is obtained by adding the increase in oxygen in the transparent, "light" bottle to the decline in oxygen in the "dark" bottle. The advantage of this method is that it partitions the water column production between daytime net production by phytoplankton and respiration by the total community. However, these measurements only capture P and R of a discreet location in the water column at a specific time and do not account for production associated with other organisms such as macroalgae and seagrasses, consumption by zooplankton and fish, and benthic respiration that occurs in the sediment. Typically, however, light and dark bottle measurements are coupled with some sort of additional measures or estimates of benthic respiration. Some weaknesses of this method are that errors associated with each measurement are additive, which amplifies the uncertainty, and there is an assumption that the bottle measurements can be scaled up to represent the whole system. While many of the earlier studies of estuarine metabolism used this method, the development of high-frequency data-logging dissolved oxygen (D.O.) and carbon dioxide (CO_2) sensors has led to a reliance on other, less sample-intensive methods.

Another way to measure metabolism in an estuary is by measuring either diel (daily cycle) or seasonal changes in D.O. Unlike the light and dark bottle method, these measurements represent whole water column changes and interaction with the benthos, integrating all sources of production and respiration, as well as the added complications associated with the physical transport of water masses and air-sea gas exchange. Whole water column concentrations of D.O. increase over the course of the day, as photosynthesis occurs, and decline at night, when photosynthesis ceases and respiration dominates. While changes in D.O. concentrations over a 24-h period can be used to calculate daily NEP, this method is not without its challenges, particularly in relation to the assumptions that changes in the dissolved gas are biologically mediated and entirely capture biological processes. In addition to accounting for exchange between the water and the atmosphere and across different water masses, some investigators have observed a significant amount of benthic respiration occurring in portions of the water and sediment columns that are devoid of O_2. Thus, the organisms respiring CO_2 are using other elements, such as sulfur, to conduct this respiration. This means that such respiration would not be captured by changes in O_2 concentrations. However, the sulfur-mediated respiration can be estimated in O_2 equivalents (Kemp et al., 1997).

Nutrient budgets

Seasonal or annual nutrient concentrations and ratios have also been used to successfully calculate TEM. For example, the ratio of dissolved inorganic nitrogen (DIN) to dissolved inorganic phosphorous (DIP) reflects the biological uptake and removal processes that use these nutrients: primary production, respiration, and denitrification. One can calculate the TEM by comparing the DIN/DIP to nutrient inputs. This technique is well illustrated by Nixon and Pilson (1984) in Narragansett Bay, Rhode Island. By comparing the annual DIN and DIP inputs to the mean annual DIN/DIP ratios in the bay, they were able to calculate that the bay is net autotrophic and production exceeded consumption by an amount of carbon equal to almost a quarter of that produced by bay phytoplankton. Similar budgeting using single nutrients can also be used. For example, Smith and Hollibaugh (1997) used DIP concentrations to estimate the NEP in Tomales Bay, an estuary in Northern California. They measured the differences between DIP inputs and outputs, where the difference between the two was used as an index of NEP. The DIP concentrations were converted to carbon units using the 106:1 Redfield ratio of DIC/DIP. In contrast to Narragansett Bay, this US West Coast system was found to be net heterotrophic. Terrestrial and marine systems each contributed equally to supporting the excess NEP in Tomales Bay.

In general, estuaries are thought to be net heterotrophic. However, on a global scale, increasing nutrient inputs from upstream may be shifting these systems to autotrophic (Kemp et al., 1997; Smith and Hollibaugh, 1997). For example, excess nutrient inputs from sewage was one of the main reasons that Narragansett Bay was autotrophic at the time of the study (Nixon and Pilson, 1984), but recent upgrades to tertiary sewage treatment may alter this regime. While Chesapeake Bay also receives substantial sewage input, most of the nutrients entering the estuary are from agricultural runoff. Kemp et al. (1997) measured NEP using five different approaches and found quite similar results in the Chesapeake. They summed all carbon fluxes (including P and R measurements made using light and dark bottles), developed nutrient mass balances, and compared nutrient ratios. In addition to strong agreement among methods, they observed clear spatial variability in the bay where the upper bay was net heterotrophic, the mid bay in balance, and the lower bay was net autotrophic. Their measured values were driven largely by the biology, not physical processes, and they concluded that the ratio of DIN to total organic carbon was the controlling factor of NEP in Chesapeake Bay, where the upper bay received more organic carbon inputs and the lower bay relied more on in situ production. Thus, measures of TEM can help to provide useful insight into the relative importance of contributions from river inputs versus production within the estuary.

Summary

Total ecosystem metabolism (TEM), both as discrete measurements and as a theoretical concept, has an important history in ecosystem ecology, particularly in estuaries.

Some of the earliest ecological studies were developed to determine how energy flowed through an ecosystem and whether these systems were net importers or exporters of organic matter. In estuaries, measurements of TEM are challenging as their dynamic nature makes them difficult to characterize. Estuarine ecologists most often focus on components of TEM, like the net ecosystem metabolism or gross primary production, which can indicate how much organic matter is consumed within an estuary or exported to the coastal ocean, as well as how impacted an estuary is by anthropogenic inputs of nutrients and organic matter.

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Cross-references

Dissolved Oxygen
Nutrients

ESTUARY CONSERVATION

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Definition

Conservation is the act of protection or maintenance over time of an ecosystem and its components. Estuary conservation can be directed to consumptive use of a living estuarine resource or nonconsumptive use such as preserving an estuary as a reserve or protected area.

Overview

Historically, conservation of estuaries and their resources were maintained by a lack of overexploitation, an absence of industrial activity that resulted in large-scale habitat disruption, and an ability of indigenous people to adapt to change. A good example is the harvest of eulachon (*Thaleichthys pacificus*), a smelt-related fish that migrates through some northeast Pacific Ocean estuaries to spawn in spring. Groups of up to 10,000 native people from nearby areas gathered at the specific estuaries to harvest the fish. The oil from the eulachon was also exported for trade inland several hundred kilometers through the mountains via "grease trails" to interior tribes (Mitchell and Donald, 2001). This estuarine resource and its habitat maintained a culture and socioeconomic system for centuries and thus were conserved (Turner and Clifton, 2009).

The conservation of estuaries and estuarine resources has become more difficult in the postindustrial era. Although ecologists have developed knowledge of how estuaries function in support of conservation, the information has not always been applied systematically through policies. The inexorable movement of people from the interior of continents to coasts and estuaries has increased harvesting pressure, and the globalization of trade with its requirement for major port development to deliver goods has created challenges. Global warming and related climate change is also a factor. Up until about 30 years ago, conservation methods were often focused on efforts to control population dynamics of single species or preserve habitats without consideration of the ecosystem.

Conservation biology, an advancing stage in the application of science to conservation problems, addresses the biology of species, communities, and ecosystems that are perturbed, either directly or indirectly, by human activities or other agents (Soule, 1985). A strong conservation plan to address problems will require careful development of a vision, goals, and objectives for the estuary. Conservation biology requires a more holistic view of the estuary and especially understanding of complex processes such as energy flow, genetic changes, maintenance of freshwater flow, and sediment transport. A comprehensive conservation or management plan for an estuary and its resources thus requires major data gathering and syntheses which can be facilitated by the use of models. As well the linkages between socioeconomics and the estuarine ecosystem are now appreciated. This is sometimes called the "comprehensive landscape approach". Tools such as the Integrated Valuation of Ecosystem Services model (InVEST) (Nelson et al., 2009) are available and are recommended to help the complex task of estuary conservation.

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Cross-references

Estuarine Sustainability
Estuary Conservation Zone
Sustainable Use

ESTUARY CONSERVATION ZONE

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Definition

An estuarine conservation zone is an area in an estuary which receives legislated protection from industrial development or other impact because of its recognized ecological value.

Overview

Criteria for conservation zones: The criteria for an estuarine conservation zone will be conditioned by local requirements and are usually decided by conservation (qv) needs as well as methods related to coastal zone management (qv). The presence of a particular endangered species, according to the criteria of the International Union for Conservation of Nature and its irreplaceability, is important (AZE, 2013). Marine or estuarine reserves (Roberts et al., 2003) are types of conservation zones that serve to protect an area for fishing in order to avoid overharvesting or to repopulate depleted fisheries. The existence of rare habitat is also important (e.g., a tract of intact habitat within a disrupted estuary).

Mapping techniques: The mapping of a conservation zone can be difficult because habitat boundaries are often not easily recognized, especially for wide-ranging estuarine animals. Conservation zones for estuarine plants are more easily delineated because methods used for estuarine vegetation mapping are fairly well described (e.g., WERC SFBE, 2013). Remote sensing is increasingly used to map conservation zones (e.g., Yang, 2009), although ground truthing is still an important task. Software is available to optimize the spatial aspects of estuarine conservation zone mapping (e.g., Marxan; see Watts et al., 2009).

Examples of estuary conservation zones: Most estuary conservation zones are described in local, regional, or national planning documents dealing with integrated coastal zone management (qv). Edwards and Winn (2006) and Edgar et al. (2010) provide case histories of the development of estuarine conservation zones.

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Cross-references

Estuarine Sustainability
Sustainable Use

EUTROPHICATION

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Synonyms

Nutrient enrichment; Organic matter enrichment

Definition

Eutrophication is a process that can severely degrade estuarine and coastal marine environments. It has been defined differently by authors in the scientific literature. Nixon (1995), for example, defined eutrophication as an increase in the rate of supply of organic matter to an ecosystem. The National Academy of Sciences (1969) defined the term as the natural or artificial addition of nutrients to a waterbody and the effects of the added nutrients on the system. Similarly, the European Union Directives has focused on nutrient inputs to a waterbody and its consequences (Anonymous, 2000). In this case, eutrophication has been defined as the process of nutrient enrichment and increase in the rate of organic matter input in a waterbody leading to an array of cascading changes in ecosystem structure and function such as decreased dissolved oxygen levels (hypoxia/anoxia), increased microalgal and macroalgal abundance, elevated epiphytic growth, occurrence of harmful algal blooms (HABs), loss of essential habitat (e.g., seagrass and shellfish beds), reduced biodiversity, declining fisheries, imbalanced food webs, altered biogeochemical cycling, and diminished ecosystem services (Kennish and de Jonge, 2011).

Eutrophication of estuarine ecosystems

Eutrophic conditions have developed in many estuarine systems bordered by watersheds with increasing agricultural and urban land use, and the effects are most acute in shallow coastal bays and coastal lagoons with restricted circulation and protracted water residence times (Nixon et al., 2001; Burkholder et al., 2007; McGlathery et al., 2007; Anderson et al., 2010; Kennish and Paerl, 2010; Giordano et al., 2011; Howarth et al., 2011). For example, moderate to high levels of eutrophication have been documented in an array of mid-Atlantic coastal lagoons (USA). Among these impacted systems are Great South Bay (NY); Barnegat Bay-Little Egg Harbor Estuary (NJ); Rehoboth, Indian, and Little Assawoman Bays (Delaware Inland Bays); Assawoman Bay, Isle of Wight Bay, and St Martin River (Northern Maryland Coastal Bays); and Newport, Sinepuxent, and Chincoteague Bays (Southern Maryland Coastal Bays). Similar impacts are apparent in other countries as well, for example, (1) Wadden Sea and Ems Estuary (Netherlands and Denmark), (2) Peel-Harvey Estuary (Australia), (3) Shenzhen Bay (China), and (4) Ghana coastal lagoons (Africa).

Sources of nutrient enrichment

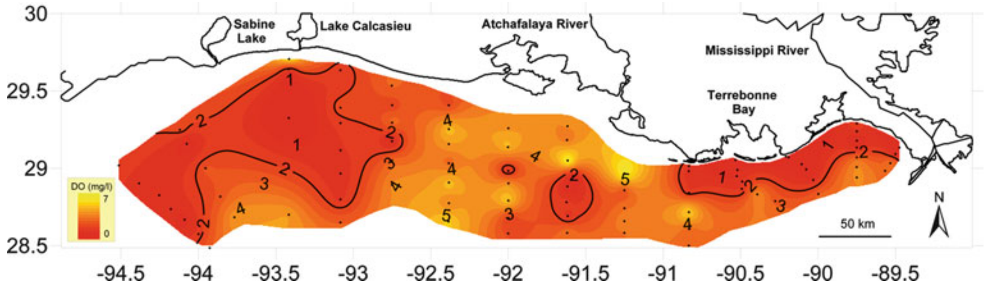
Human population growth and development continue to escalate in the coastal zone. More than 75 % of the world's human population inhabits coastal river basins. Hence, it is understandable why nutrient enrichment has become a major problem in estuaries worldwide. Major sources of nutrients to estuarine systems include farmlands, stormwater runoff, wastewater discharges, groundwater seepage, and atmospheric deposition. Anderson et al. (2010) stressed that nutrient enrichment by reactive nitrogen in coastal waters is accelerating due to development and intensification of agriculture. Nonpoint sources of nutrient pollution are significant not only due to increased development, greater impervious surfaces, and runoff in coastal watersheds but also due to atmospheric deposition. According to Paerl et al. (2002), for example, atmospheric deposition accounts for ~10–40 % of the new nitrogen loading to estuaries investigated along the East Coast and eastern Gulf of Mexico (USA). Complicating matters, estuaries are important repositories of nutrients delivered from watersheds and the atmosphere. For example, bottom sediments typically serve as a pool and a secondary source of nutrients to the water column in these susceptible ecosystems. Studies have shown that nutrient concentrations in estuarine bottom sediments may be 10- to 100-fold higher than in the water column (Sand-Jensen and Borum, 1991; Burkholder et al., 2007).

Ecosystem impacts

Nutrient enrichment (most notably nitrogen and phosphorus) from coastal watersheds and the atmosphere (via dry and wet deposition) is an important driver of biotic change in estuaries. It can cause significant shifts in primary production and plant biomass, as well as changes in the composition of autotrophs – microalgae, macroalgae, and rooted macrophyte assemblages that modulate higher trophic-level dynamics. Thus, the effects of altered bottom-up controls on the biotic structure and function of a system can be far reaching. Nutrient enrichment and the resulting eutrophic impacts often pose serious threats to an estuary because they can lead to long-term, ecosystem-wide decline, adversely affecting biotic resources and human uses.

Nitrogen concentrations in estuarine waters typically range from <1 to 60 μmol l^{-1}, while phosphorus concentrations generally range from <1 to 10 μmol l^{-1} (Kennish, 2001a). Much higher nutrient levels may occur in eutrophic systems. Phosphate concentrations are usually much lower than nitrate concentrations in estuarine waters in large part because phosphate readily sorbs to particulate matter or forms insoluble precipitates that accumulate in bottom sediments (Kennish and de Jonge, 2011). This process affects the biologically available fraction of phosphorus. Both inorganic and organic nutrient forms must be considered in assessment of estuarine eutrophication.

Depending on the physicochemical and biotic conditions, the consequences of nutrient enrichment and increase in the rate of organic matter supply in an estuary are numerous and varied which can cause declining system stability and resilience. The long-term effect of eutrophication is potentially the permanent alteration or loss of biotic communities and habitats and great ecosystem-level degradation. This is so because the eutrophication process disrupts the ecological interrelationships and functioning of coastal water bodies. Nutrient enrichment stimulates algal production and sets into motion changes in the ecosystem from the bottom-up, literally altering the foundations of the way the estuary functions. As algal populations bloom, die off, and then sink to the floor of



Eutrophication, Figure 1 Bottom dissolved oxygen concentrations in the northern Gulf of Mexico (USA) during summer 2010. Note latitude and longitude expanse of hypoxic areas (Illustration courtesy of Nancy N. Rabalais, Louisiana Universities Marine Consortium, Chauvin, Louisiana).

the estuary, they undergo microbial decomposition which uses up oxygen, causing stress and loss of living resources.

Dramatic increases in nutrient enrichment of coastal waters have occurred over the past several decades driven by anthropogenic activities (Kennish, 1992; Kennish, 1997; Kennish, 2001a; Kennish, 2002; Kennish and de Jonge, 2011). Howarth et al. (2002), for example, showed that the amount of dissolved inorganic nitrogen transported from rivers to coastal ocean waters nearly doubled between 1961 (3.0 Tg-N year^{-1}) and 1997 (5.0 Tg-N year^{-1}), reflecting increased fertilizer use, fossil fuel combustion, and nitrogen fixation in agricultural systems (see Galloway et al., 2002). Howarth et al. (1995) also revealed that phosphorus loads through the river-estuary-ocean continuum have increased markedly from historic levels of $\sim 8 \times 10^6$ mt (metric tons) year^{-1} to $\sim 22 \times 10^6$ mt year^{-1}.

While nitrogen is generally regarded as the primary nutrient of concern in eutrophic estuaries, phosphorus must also be considered because it can contribute significantly to eutrophication and may be the principal control of primary production in some estuaries. Hence, nutrient pollution abatement should focus on reduction of both nitrogen and phosphorus loading to estuarine waters (Conley et al., 2009).

Hypoxia (low dissolved oxygen in the water, <2 mg l^{-1}) and anoxia (dissolved oxygen concentrations $= 0$ mg l^{-1}) of estuarine and coastal marine environments are increasing with escalating nutrient enrichment (Diaz and Rosenberg, 1995; Kennish, 2002; Diaz and Rosenberg, 2008; Kennish and Paerl, 2010). Increasing nutrient inputs to these environments leads to greater organic matter production, biomass accumulation in bottom sediments, higher biochemical oxygen demands, and accelerated microbial decomposition of the accumulated detrital biomass which depletes oxygen concentrations, with adverse effects resonating through higher trophic levels. This process is evident in some highly stratified water columns such as Chesapeake Bay, which has a long history of seasonal declines in deepwater dissolved oxygen concentrations (Boynton and Kemp, 2000). Other examples are the northwestern shelf of the Black Sea, the large coastal areas of the Baltic Sea with expansive hypoxic zones, and the Louisiana inner shelf in the northern Gulf of Mexico (USA) (Rabalais et al., 2007; Kennish and de Jonge, 2011). Figure 1 illustrates the broad expanse of hypoxic waters in the northern Gulf of Mexico in 2010.

An accurate method of assessing eutrophic conditions in estuaries is the use of bioindicators together with physical and chemical water quality indicators. Bricker et al. (2007) assessed conditions and trends of eutrophic symptoms within US estuaries using this approach. They found that 65 % of the assessed estuarine area in the USA (64 of 99 estuaries examined) had moderate-to-high eutrophic levels. Coastal lagoons were the most heavily impacted by eutrophication. Furthermore, Bricker et al. (2007) noted that eutrophic conditions are expected to worsen in many of these estuaries by 2020.

Significant biotic changes occur in eutrophied systems. For example, the species composition, abundance, distribution, and diversity of organisms commonly change in these systems, including primary producers and top-down feeding groups that regulate algal populations, keeping them in check. Essential plant habitat (i.e., seagrass) is commonly replaced by less desirable nuisance algae (e.g., macroalgae, *Enteromorpha* spp., and *Ulva* spp.) which outcompete the vascular plants (Burkholder et al., 2007; McGlathery et al., 2007; Kennish et al., 2010). A positive correlation exists between nutrient loading and algal production and biomass. The accumulation of large amounts of decaying algae on the estuarine floor not only promotes hypoxic conditions but also the production of sulfides in bottom sediments, mediated by microbial activity that can be extremely toxic to bottom-dwelling communities. Phytoplankton and macroalgal blooms, epiphytes, and suspended particulates also attenuate or block light transmission to seagrass beds that can cause dieback and elimination of this bottom habitat for fish and shellfish.

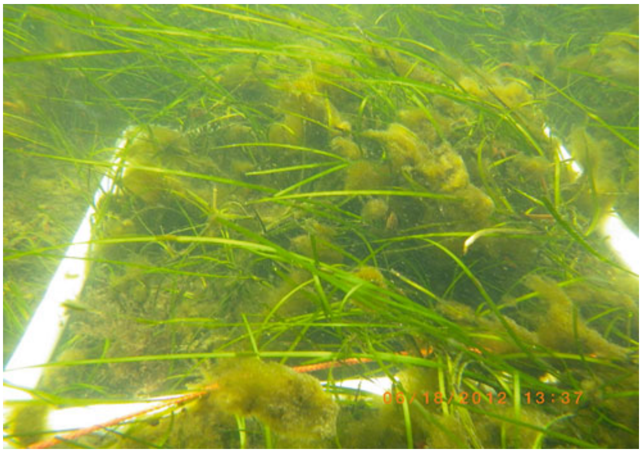
Additional biotic impacts associated with nutrient enrichment include changes from filter-feeding to deposit-feeding benthic invertebrates. Progressive change occurs in benthic invertebrate communities of many eutrophic systems from larger, long-lived benthic infauna (e.g., hard clams, *Mercenaria mercenaria*) to smaller, rapidly growing, but shorter-lived forms (e.g., coot clams, *Mulinia lateralis*). The loss of larger, filter-feeding shellfish species has been well documented in eutrophic systems, such as the Barnegat Bay-Little Egg Harbor Estuary in New Jersey (USA), which reduces bottom-up control and regulation of phytoplankton populations (Kennish, 2001b). In an environment of less top-down control, toxic phytoplankton blooms may also occur (e.g., brown tide, *Aureococcus anophagefferens*). The potential for permanent alteration of biotic communities and habitats exists in this type of impacted system, and its stability and resilience are likely compromised.

Barnegat Bay-Little Egg Harbor Estuary: case study

Barnegat Bay-Little Egg Harbor (BB-LEH) Estuary has been classified as a highly eutrophic coastal lagoon based on application of NOAA's National Estuarine Eutrophication Assessment (NEEA) Model (Bricker et al., 2007) and Nixon's Trophic Classification (Kennish et al., 2007; Kennish et al., 2010; Fertig et al., 2013). This 280-km^2 estuarine waterbody is highly susceptible to nutrient loading because it is shallow, poorly flushed, and bordered by highly developed and altered watershed areas (1,720 km^2) that act as a conduit for nutrient transport to the estuary. Nutrient enrichment in this waterbody, as well as other coastal lagoons in the mid-Atlantic region, is linked to an array of adverse impacts, most notably eutrophication.

Total nitrogen loading from the BB-LEH watershed ranges from ~455,000 to 857,000 kg N year^{-1}; total phosphorus loading in turn ranges from ~17,000 to 32,000 kg P year^{-1} (Baker et al., 2013). The nitrogen loading is a major driver of ecological change in the estuary (Fertig et al., 2014). Highest loading occurs in the northern segment of the estuary in closest proximity to the most highly developed areas of the watershed. Elevated total nitrogen levels have been detected in the north and south segments of the estuary (Fertig et al., 2013).

Studies of coastal lagoonal systems indicate that environmental impacts escalate as development and the amount of impervious cover in surrounding coastal watersheds increase. A watershed impact threshold is exceeded when the amount of impervious surface cover is greater than 10 % (Arnold and Gibbons, 1996). Development of the BB-LEH watershed now amounts to ~34 %, and the impervious land cover exceeds 10 %. Ecological impacts therefore are to be expected with increasing land alteration in the watershed (Lathrop and Conway, 2001; Kennish et al., 2007; Fertig et al., 2014). The BB-LEH Estuary is an ecologically impacted system. This is manifested by declining ecological conditions such as significant loss of seagrass, occurrence of nuisance and toxic algal blooms (including brown tides), heavy epiphytic loading, markedly diminished fisheries (e.g., hard clams, *Mercenaria mercenaria*), eruptions of deleterious organisms (e.g., sea nettles, *Chrysaora quinquecirrha*), decreasing biodiversity along hardened shorelines (which now cover 40–45 % of the estuarine shoreline), and other degrading changes. These adverse effects have become increasingly evident during the past 15 years. Extensive studies, peer-reviewed publications (including references therein), and numerous technical reports published on the estuary during the past two decades have clearly documented these problems (Seitzinger et al., 1993; Bricker et al., 1999; Bologna et al., 2000; Kennish, 2001a; Lathrop and Bognar, 2001; Seitzinger et al., 2001; Gastrich et al., 2004; Bricker et al., 2007; Kennish and Townsend, 2007; Kennish et al., 2007; Kennish et al., 2008; Kennish, 2009; Moore, 2009; Kennish et al., 2010; Kennish et al., 2011; Lathrop and Haag 2011; Kennish and Fertig, 2012; Fertig et al., 2013).



Eutrophication, Figure 2 Macroalgal bloom in a seagrass bed of the Barnegat Bay-Little Egg Harbor Estuary (USA) in June 2012.

Nutrient enrichment elicits negative biotic responses in BB-LEH. For example, nitrogen loading stimulates algal growth and epiphytic infestation that cause light attenuation and shading of seagrasses (Kennish, 2001b; Kennish et al., 2011; Fertig et al., 2013, 2014). Blooms of drifting, ephemeral macroalgae (e.g., *Ulva lactuca*, *Enteromorpha intestinalis*, *Gracilaria tikvahiae*, and other species) have produced thick canopies of organic matter that pose a potential danger to the seagrass beds by smothering the plants and blocking light penetration (Kennish et al., 2007, 2008, 2011; Kennish and Fertig, 2012). Figure 2 shows a macroalgal bloom in a seagrass bed of the estuary in June 2012. Additionally, the accumulation of these macroalgal mats on the estuarine floor can cause an increase in sediment sulfide concentrations due to microbial decomposition in anoxic, organic-rich sediment layers that is detrimental to seagrasses and benthic infaunal communities (Burkholder et al., 2007; Anderson et al., 2010). Seagrass photosynthesis, metabolism, and growth are negatively affected by sulfide buildup in bottom sediments leading to a decrease in the depth penetration of seagrasses in eutrophic waters (National Research Council, 2000; Burkholder et al., 2007; McGlathery et al., 2007).

In BB-LEH, comprehensive investigations of seagrass beds have documented significant declines in plant demographics (biomass, blade length, and areal cover) during the 2004–2010 time period in response to nitrogen inputs (Kennish et al., 2008; Kennish et al., 2010). For example, after an extended progressive decline, *Zostera marina* (eelgrass) mean aboveground and belowground biomass values in 2010 were the lowest ever recorded in the estuary (7.5 and 26.7 g dry wt m^{-2}, respectively), which were 87.3 % and 64.8 % lower than in 2004. Concurrently, mean *Z. marina* areal cover decreased by 43.8 %, and mean blade length declined by 33.7 %. In addition, mean shoot density of *Z. marina* during 2004–2010 (~240–495 shoots m^{-2}) also decreased substantially compared with earlier measurements (650–1,150 shoots m^{-2} in 1999, Bologna et al., 2000, and 500–1,000 shoots m^{-2} in 1982, Vaughan, 1982).

Concomitant with the loss of seagrass habitat over the 2004–2010 period, other impacts have been documented. For example, macroalgal blooms increased during 2004–2010 as well; investigations of macroalgal blooms in the estuary over this period revealed 55 occurrences (2.23 blooms m^{-2}) of early bloom (70–80 % macroalgal cover) and full bloom (>80 % macroalgal cover) events (Kennish et al., 2011). These resulted in increased mortality of seagrass leading to reduced biomass and bare-bottom areas within the beds. Furthermore, epiphytic overgrowth on seagrass was elevated, with the mean percent cover of seagrass leaf surfaces ranging from 10.7 % to 38.3 % during 2009 and 2010. The loss of seagrass habitat has eliminated essential habitat for hard clams, bay scallops (*Argopecten irradians*), and other benthic and demersal organisms. Seagrass now covers a 5,260-ha area of the BB-LEH estuarine floor, but the biomass of the seagrass beds is now significantly reduced (Kennish et al., 2010; Fertig et al., 2013; Fertig et al., 2014).

The decline of seagrass beds is a serious concern in any estuary because of the multiple ecosystem services that they provide, notably major sources of primary production, food for waterfowl, essential habitat and nursery areas for numerous fish and invertebrates, filters of chemical substances, agents in biogeochemical cycling, and buffers against wave and current action as well as sediment erosion (Larkum et al., 2006; Orth et al., 2006; Moore, 2009). These vascular plants are important indicators of overall ecosystem health of an estuary because they integrate water quality and benthic attributes (Longstaff and Dennison, 1999; Carruthers et al., 2002; Orth et al., 2006; Burkholder et al., 2007; Kennish et al., 2008; Kennish et al., 2010; Moore, 2009).

Management of eutrophication

The conversion of natural land covers to farmlands, housing developments, and industrial complexes facilitates nutrient loading to nearby estuarine waters, leading to cascading water quality, biotic, and habitat impacts as well as diminished ecosystem services. Natural stressors, such as hurricanes and other major storms as well as floods and droughts, can exacerbate these effects (Paerl et al., 2005; Paerl et al., 2007; Paerl et al., 2009). An array of mid-Atlantic estuaries of the USA, most notably coastal lagoons with restricted circulation and high water residence times, has exhibited severely stressed responses due to nutrient over-enrichment. Most lagoonal estuaries in this region are now moderately to highly eutrophic and rank among the most impacted estuarine systems in the USA (Bricker et al., 1999; Bricker et al., 2007). Watershed management strategies to reduce nutrient loading in estuaries of this region include upgrading stormwater controls, implementing low-impact development and best management practices, advancing open space preservation, and generating total maximum daily loads (TMDLs) for nutrient limitation. The effects of urbanization as a driver of nutrient impacts in estuaries will continue to increase with increasing coastal population growth, development, and alteration of coastal watersheds, unless aggressive management actions and effective land use planning are broadly implemented.

There is the need to scale up the magnitude of corrective actions in altered coastal watersheds where estuaries are impaired from nutrient enrichment. The effort to protect an estuary from ongoing nutrient pollution must be accelerated. Coastal population growth and development have altered land use in the coastal zone and contributed significantly to the ecological downspiral of many estuaries. Both point and nonpoint source inputs of nutrients to estuaries continue to be problematic in both developed and undeveloped countries, and the future projections for improvement are not promising. In addition, engineering controls have not been effective at correcting the serious environmental impacts from the cumulative land use-land cover changes in many coastal watersheds. More aggressive management strategies and decisions are needed to effectively address continued development effects in watersheds, if the long-term remediation of estuarine eutrophication is to be achieved.

Another problem that must be effectively addressed is atmospheric delivery of nutrients to estuaries. Atmospheric deposition of nitrogen on estuary and watershed surfaces is substantial and detrimental (Paerl et al., 2002). Nitrogen inputs from the atmosphere derive substantially from fossil fuel combustion, and it is important to seek solutions for this problematic source.

Summary

Boesch et al. (2001) discussed the strategies used to control nutrient pollution in estuarine and marine systems. They identified three principal strategies of nutrient pollution abatement: (1) nutrient control at the source (i.e., farms, animal feedlots, lawns, and fossil fuel power plants), (2) reduction of nutrient use, and (3) nutrient removal during transport to the sea. All are difficult to achieve.

No single management control will likely resolve eutrophication problems in an estuary. Indeed, the most successful management strategies include a holistic environmental approach to reduce nutrient loading. It is also vital to address population growth and development in coastal watersheds that have resulted in substantial land use-land cover changes and recalcitrant ecological impacts in estuaries. Equally important is the need to implement best land use management practices in watersheds, preserving open space, initiating restoration efforts, and educating the public as to how and why these strategies are important and necessary to protect estuarine systems. Establishing total maximum daily loads (TMDLs) for nutrients may be necessary to remediate eutrophication in heavily impacted systems.

Eutrophication can severely impair an estuarine waterbody. If eutrophication problems are not expeditiously addressed in heavily impacted systems, they may become intractable, and the long-term ecological health of these systems can be permanently compromised (Kennish and de Jonge, 2011). Estuaries subjected to eutrophication through time may exhibit nonlinear recovery trajectories and changes in stable states such that they cannot return to the original ecosystem condition after the eutrophic stressor is removed. The net effects of long-term and progressive eutrophication are often substantially degraded biotic and habitat components of an estuary.

It is becoming increasingly clear that construction in coastal watersheds should minimize the creation of impervious surfaces, compacted soils, and sprawl, while concurrently preserving natural vegetation and landscapes that will assimilate nutrients. This approach will limit the amount of stormwater and pollution runoff to area waterways. These strategies make sound environmental sense not only to help remediate nutrient enrichment of estuaries but also to address other nonpoint source pollution problems that threaten estuarine ecosystem health. Application of these strategies must proceed even as more assessment data are being compiled by ongoing research and monitoring efforts to document ecosystem change and to recommend future impact remediation. A well-coordinated and sustained holistic management plan is critical to improving the ecological condition and resources of eutrophic estuaries worldwide.

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EVAPORATION AND TRANSPIRATION

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Definition

Evaporation. The physical process whereby water as liquid or moisture is converted to vapor.

Transpiration. The process whereby water is evaporated from the leaves, stems, and flowers of a plant.

Description

Evaporation and transpiration result in the export of water as vapor from estuaries and their coastal environments and tidal flats, causing water loss, desiccation, salinization, and physiological stress of biota. The effect of evaporation on estuaries and their peripheral tidally exposed environments can be substantial in regions with high solar-induced evaporation and those with strong coastal winds. In tropical arid climates, high evaporation (e.g., 3,000 mm/pa) and strong coastal winds (>20–30 km/h) commonly occur together.

Estuarine water bodies, their coastal zones, and tidal flats are subject to the evaporative effects of solar radiation and wind. Evaporation of a shallow estuarine water body over a year (or years), if there is little recharge from rivers and microtidal conditions, will concentrate salt. At extremes, evaporation results in periodic hypersalinity in the upper estuary (an inverse estuary).

Evaporation in coastal habitats and tidal flats will result in salinization of sediments and physiological stress of biota (by direct desiccation and by salinization). This is particularly so for tidal flats as they tend to be relatively wide with laterally extensive surfaces exposed to sun and wind, providing large surface areas for moisture loss. During low tide, water under exposed tidal flats occurs as a shallow water table (the top of a phreatic zone that is contiguous with and tidally oscillating with open estuarine waters) and as pore water (a vadose zone in the sediment above the water table) – evaporation causes their salinity to increase. With a low-gradient sloping surface, tidal flats are graded with respect to inundation, and salinity thus increases upslope because of the progressively longer exposure to evaporative processes, where groundwater and pore water salinity may reach 100–150 ppt. This gradient of salinity underpins the biological zoning on the tidal flat as each species responds to the various levels of inundation and salinity. Evaporation directly desiccates tidal flat biota, physiologically stressing them, and this process also determines biological occurrence and zonation. Increase in salinity of groundwater and pore water, in addition, may progress to the point that minerals such as calcite and gypsum precipitate in sediments as crystals, layers, or nodules.

For transpiration, since most of the plants that transpire are tidal (samphires, reeds, rushes, sedges, or mangroves), the most marked water vapor loss by this process is on vegetated tidal flats. Transpiration draws on pore water and shallow groundwater, thus concentrating salt and augmenting the salinization of tidal flat induced by sun and wind. The rates of transpiration for saltmarsh and mangroves are variable across species and related to the size and density of plants, season (generally higher in the dry season than the wet), time of day, and salinity of the tidal flat. For plants of the saltmarsh, transpiration can vary from ~1 to 6 mmol $m^{-2}s^{-1}$; for mangroves, they vary from 0.5 to 7.0 mmol $m^{-2}s^{-1}$ (and generally 2–4 mmol $m^{-2}s^{-1}$) (Adam, 1990; Kathiresan and Bingham, 2001; Saenger, 2002). As such, transpiration can be a significant factor in moisture loss in vegetated peripheral estuarine environments.

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Cross-references

Mangroves
Saltmarshes
Tidal Flat
Tidal Flat Salinity Gradient

EXOTIC SPECIES

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Synonyms

Alien species; Introduced species; Non-native species

Definition

Species introduced into an ecosystem (e.g., an estuary) where they do not occur naturally.

An exotic species, with respect to a particular ecosystem, refers to any species, including its larvae, seeds, eggs, spores, or other biological material capable of propagating the species, which is not native to that ecosystem (Beck et al., 2008). An exotic species does not necessarily cause economic or environmental harm to an ecosystem.

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Cross-references

Introduced Species
Invasive Species

EXTRATROPICAL STORMS

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Synonyms

Extratropical cyclone; Midlatitude cyclone

Definition

Extratropical storms are cyclones that originate in mid to high latitudes, discernible by a distinct front due to the contrast in temperature between adjacent air masses.

Extratropical cyclones are low-pressure systems that originate outside of the tropics, resulting from the interaction of warm and cold air masses. The center of these storms contains the lowest pressure and is colder than the surrounding storm system. In the Northern Hemisphere, winds rotate counterclockwise (opposite in the Southern Hemisphere), pulling and mixing warm air from the tropics with cold air from northern latitudes. The boundary where warm and cold masses meet is termed a "front." The resulting front produces severe weather with extensive precipitation, with the strongest winds located in the upper atmosphere. Extratropical storms generally move west to east (in the Northern Hemisphere) across the globe and are the dominant storm system for much of fall, winter, and spring outside of the tropics (NHC, 2013).

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Cross-references

Extreme Events (Hurricanes)
Storm Surges

EXTREME EVENTS (HURRICANES)

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Synonyms

Cyclone; Tropical cyclone; Typhoon

Definition

Tropical cyclones are storms with a minimum sustained surface wind speed of 74 mph (64 kt) in the Atlantic and eastern Pacific Oceans. The sustained wind speed is the average wind speed in a 2-minute period.

Hurricanes are a type of tropical cyclone. Tropical cyclones are low-pressure weather systems that develop over warm, tropical, or subtropical waters. A tropical cyclone with sustained wind speeds of 38 mph (33 kt) or less is termed a "tropical depression," between 39 and 73 mph (34–63 kt) a "tropical storm," and above 74 mph (64 kt) a "hurricane." Hurricanes are classified according to the Saffir–Simpson Hurricane Scale. A Category 1 storm with winds between 74 and 95 mph (64–82 kt) will produce some damage; a Category 2 storm with winds between 96 and 110 mph (83–95 kt) will cause extensive damage; a Category 3 storm with winds between 111 and 129 mph (96–112 kt) will cause devastating damage; a Category 4 storm with winds between 130 and 156 mph (113–136 kt) will cause catastrophic damage; and a Category 5 storm with winds of 157 mph or higher (137 kt or higher) will cause catastrophic damage. Extreme tropical cyclones are called "hurricanes" in the Atlantic and eastern Pacific Oceans, "typhoons" in the Western Pacific, and "Cyclones" in the Indian Ocean (NOAA, 2013a). The Atlantic Ocean hurricane season begins June 1 and ends November 30th, while the Eastern Pacific season begins May 15th and ends November 30th (NOAA, 2013a).

Hurricanes have winds that rotate counterclockwise (in the Northern Hemisphere) and inward towards the storm's center where the strongest winds are located near the earth's surface. The center or core of a hurricane is termed the "eye." The eye is the warmest part of a hurricane, relatively calm, thunderstorm-free, and surrounded by an "eyewall." The eyewall contains the strongest winds of the hurricane. Hurricanes can vary in size as rainbands, consisting of high winds and thunderstorms extending outward from the hurricane's eye (NOAA, 2013b). Hurricanes may produce microbursts, tornadoes, flooding from excessive rainfall, and severe storm surge and waves (FEMA, 2013). In the Northern

Hemisphere, the most dangerous area of a hurricane is typically the northeastern quadrant, with the strongest winds and largest storm surge due to the counterclockwise rotation of the winds and forward speed of the storm.

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Cross-references

Extratropical Storms
Storm Surges

FIDDLER CRABS

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Definition

Fiddler crabs are small (up to ∼5 cm across) ocypodid crabs found worldwide in temperate and tropical estuaries. They live along bay beaches and brackish intertidal salt marshes, mangroves, mud flats, lagoons and swamps.

Description

There are about 100 named species of fiddlers, all in the genus *Uca* (Figure 1). They are semiterrestrial and live in the intertidal zone where they dig burrows, which are used for protection. The name "fiddler" is probably derived from the enlarged claw of males, which they wave to attract females during the mating season and to warn intruders who come too close to their burrow. Unlike most intertidal organisms, they are active in the air during low tide and inactive in their burrows during high tide. They feed by processing sediments and eating detritus and microalgae in the sediments. Their behaviors have been the subject of considerable study by biologists. Like all crabs, they molt in order to grow. If they have lost legs or claws, they can regenerate them and the new one will unfold when they molt. Newly molted crabs with soft shells are vulnerable and generally hide in their burrows until their new shell hardens.

Fiddler Crabs, Figure 1 Courtesy Wikimedia.

Cross-references

Blue Crabs
Soldier Crabs (Mictyridae)

M.J. Kennish (ed.), *Encyclopedia of Estuaries*, DOI 10.1007/978-94-017-8801-4,

FIRTH

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Synonyms

Flooded glacial valley

Definition

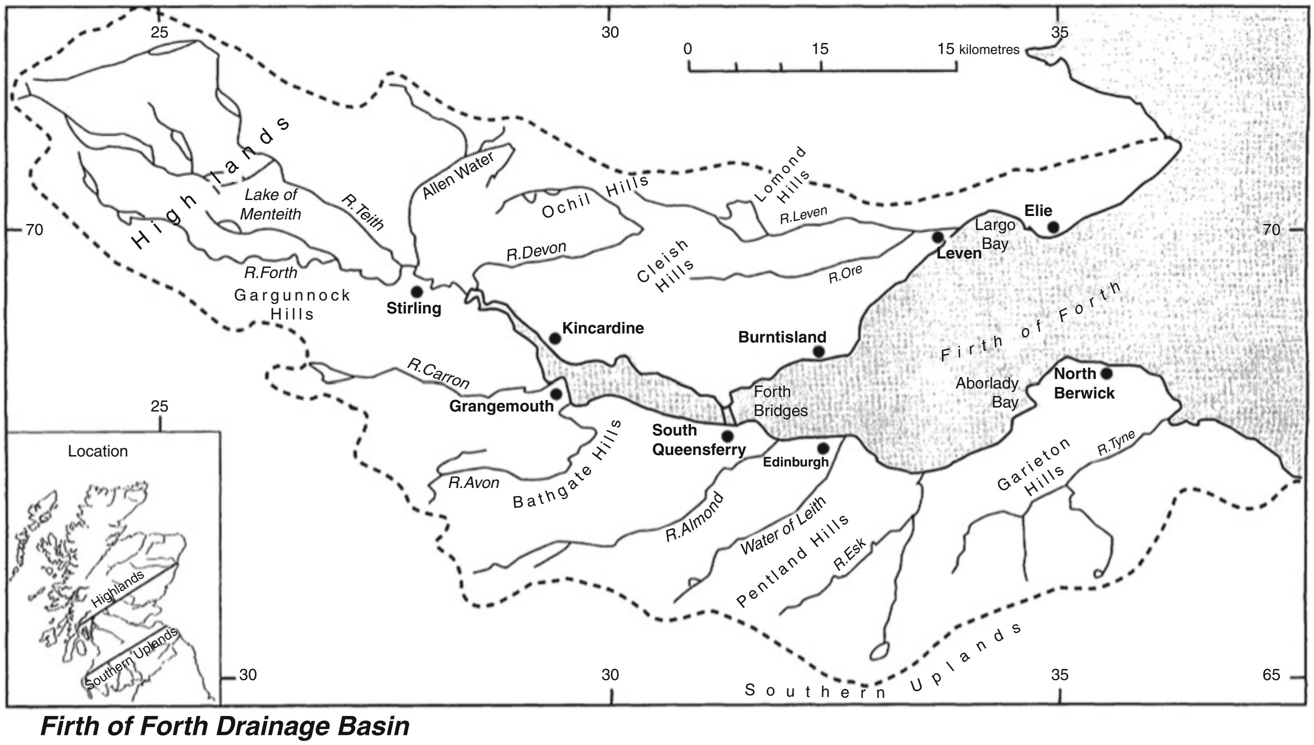
The term *firth* is a modification of the more general term *fjord* and is used to refer to the estuary formations along Scotland's eastern coast. Firths are shallower, longer, and more heavily silted than fjords. They also act as sediment traps for littoral drift and show evidence of significant infilling (Firth et al., 1997). There are strong analogies with the term *bay* used for other geographical areas of glacial origins, such as along the northeast coast of North America.

Genesis

Both the terms *firth* and *fjord* are generally applied to water bodies resulting from the flooding of glacial valleys. In general, intrusion of the sea into glacial valleys stems from two factors: Quaternary eustasy and tectonic sinking of grabens along the fractured plate margin borders. The latter is greater on the western facade. Isostatic rebound due to the present interglacial period seems to have had a greater effect on firths. There is a conceptual symmetry with the term *ria*, which also refers to coastal flooding but of river valleys. Both kinds of estuaries have equivalent characteristics but different typologies (McManus et al., 1993; Duck et al., 1995).

The flooded valleys of both firths and fjords are created and carved out by glacial action, which produces their depth and breadth; these features are also evident in subaerial segments, even if the rivers that drain them are subsequently able to rejuvenate the relief. Some show a certain hydrological hierarchy of glacial origin that endows the coast with its typical irregular morphological and multi-lobed nature. In general, they receive river flows at their heads. Some that formed more recently are steeper, but they are always limited in length.

Glacial flows have a huge abrasive and erosive effect on the sides and floors of valleys. Once the glaciers retreat, both their moraine deposits and the valley's abraded surfaces are highly prone to weathering and erosion, which means that the supply of sediment to the outlet can be enormous.



Firth, Figure 1 Firth of Forth Drainage Basin.

These effects are much more pronounced in the longer valleys of Scotland's eastern coast, with their limited number of firths, than in the shorter and much deeper valleys of the narrower western coasts with their extraordinary proliferation of fjords.

The current hierarchy of the water network is dictated by glacial erosion since the magnitude of the current interfluvial structures and the proximity of the heads of glaciers to the coast have limited the transformative action of fluvial geodynamic activity (Figure 1). Within these common traits, firths are further characterized by longer and more structured fluvial networks, greater abundance of fluvial sediment, and much smaller average gradient in the valleys as a whole. These factors result in shallower depths all along the basin, even in the coastal platform, where the oldest glacial deposits have been incorporated in littoral processes to accentuate the current silting processes in the firths.

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Cross-references

Coastal Bays
Fjord
Ria

FISH ASSEMBLAGES

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Definition

Fish Assemblage. A fish assemblage is simply a suite of species whose individuals are collected in the same area at the same time.

Introduction

There are no strict limits on the spatial or temporal scale of a fish assemblage (Miller, 2002). It may be an assemblage in a single season and region (e.g., a fish assemblage of an estuarine main channel during the late rainy season) or at specific times and locations (e.g., fish assemblages among different estuaries in South America). Research on fish assemblages has sought to understand where, why, and when they occur and how these spatial and temporal distributions relate to patterns in reproduction, recruitment, and foraging. The seasonal variation in proportion and abundance of freshwater and marine fish species in fish assemblages in estuarine habitats is a result of both the seasonal variation of environmental gradients (e.g., salinity, temperature, turbidity) and a suite of biological variables, including reproduction and recruitment (Blaber, 2000; Barletta et al., 2005, 2008; Dantas et al., 2010, 2012). In tropical estuaries, seasonal fluctuations of salinity are a major factor determining larval abundance (Barletta-Bergan et al., 2002a; Barletta-Bergan et al., 2002b), juvenile and adult density, and biomass of fishes (Blaber, 2000; Barletta et al., 2005, 2008), mainly due to the effects of large fluctuations of freshwater inputs during the year. Factors such as geology, geomorphology, and more immediate environmental factors as salinity and temperature affect fish distributions, species richness, and fisheries catch (Mahon et al., 1998; Mathieson et al., 2000; Araújo and Azevedo, 2001; Roy et al., 2001; Thiel et al., 2003). In addition, broadscale comparisons of tropical estuaries across zoogeographic realms indicate possible significant differences in the fish assemblage use of various estuarine habitats, as well as differences in the patterns of species composition (Blaber, 2000). For example, the relative proportions of freshwater and marine species using estuaries may differ (Blaber, 2000; Barletta et al., 2003, 2005, 2008). An understanding of the roles and relative importance of different habitats within estuaries to fish is necessary for both effective management and conservation (Elliott and Hemingway, 2002).

Estuarine fish guilds

The fish assemblages inhabiting estuaries worldwide have common features (Blaber, 2000; Elliott and Hemingway, 2002; Barletta and Blaber, 2007). Estuaries are nursery grounds, migration routes, and refuge areas for many fish species, including resident fish assemblages (Blaber, 2000; McLusky and Elliott, 2004; Barletta et al., 2010; Dantas et al., 2012). The study of estuarine fish assemblage structure and functioning is important for understanding the ecological characteristics of estuaries, as well as for classifying species for use in management of anthropogenic impacts in estuaries (McLusky and Elliott, 2004; Elliott et al., 2007).

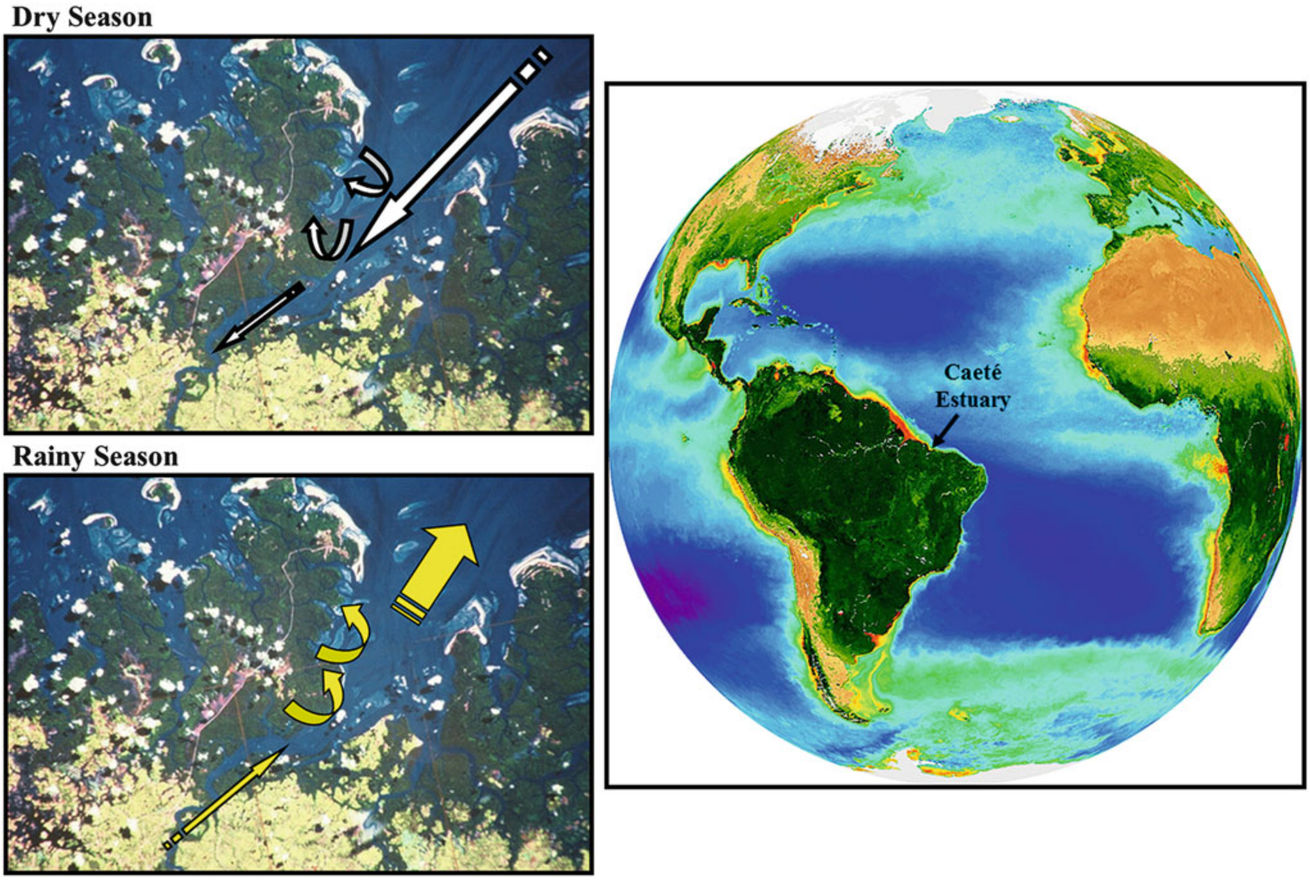
Recently, studies have concentrated on the functional analysis of finfish assemblage structure in which the species present are assigned to groupings or guilds, each of which denotes certain attributes (Elliott et al., 2007). A guild is a group of species that exploit the same class of environmental resources in a similar way (Root, 1967). Hence, guilds have been used to provide information on functioning, hierarchical structure, and connectivity and to simplify complex ecosystems (McLusky and Elliott, 2004; Barletta and Blaber, 2007). While there are many options for functional guilds, such as one for habitat preference and another relating to position inhabited within the water column (Elliott and Dewailly, 1995), Elliott et al. (2007) proposed three groups of functional guilds. The first is the Estuarine Use Functional Group, which defines the overall ecological use of an estuary by a given species and is composed by eight categories: marine stragglers, marine migrants, estuarine species, anadromous, catadromous, amphidromous, freshwater stragglers, and freshwater migrants. The second is the Feeding Mode Functional Group, which defines the primary method of feeding used by a given species and is composed by seven categories: planktivorous, detritivorous, herbivorous, piscivorous, benthophagous, hyperbenthophagous, and opportunistic. The third is the Reproductive Mode Functional Group that indicates how, and in some cases where, an estuarine species reproduces, being composed of three main categories: viviparous, ovoviviparous, and oviparous. To explore possible fundamental similarities and differences in estuarine fishes in tropical, subtropical, and temperate estuaries from different zoogeographic regions, it is necessary to compare their fish assemblages, not only taxonomically, but also in terms of their ecological structure and resource use in different habitats of the estuary during different time scales. Based on this information, Barletta and Blaber (2007) made a comparison between Caeté (Northern South America) and Embley (Northern Australia) estuaries. They found important taxonomic differences between the two estuaries. The Neotropical fish species of the upper Caeté Estuary have no equivalents in the Embley Estuary, and the diverse chondrichthyan fauna of the Embley have no equivalents in the Caeté Estuary. On the other hand, the more ubiquitous families Engraulidae, Sciaenidae, Ariidae, Carangidae, Haemulidae, and Clupeidae, which were characteristic of the main channel and mangrove tidal creeks of the Caeté Estuary, showed 70 % similarity with the main channel of the Embley Estuary.

Environmental influence on fish assemblages

Estuaries are important ecosystems for marine fisheries, and diverse authors have emphasized that fish landings around the world consist of species that spend part of their lives in estuarine waters (Blaber, 2000; Barletta et al., 2003, 2005, 2008; Barletta-Bergan et al., 2002a; Barletta-Bergan et al., 2002b). Estuarine and shallow marine waters in tropical (Blaber, 2000; Barletta et al., 2005), subtropical (Jaureguizar et al., 2004; Barletta et al., 2008), and temperate (Thiel et al., 1995) regions are important areas for feeding, mating, spawning, and nursery habitat for many fish species. The species composition of estuarine fish assemblages is dictated by a combination of biotic and abiotic variables, particularly competition for space and food, and tolerance to diel and seasonal changes in salinity, turbidity, and temperature (Blaber, 2000; Barletta et al., 2003, 2005, 2008). Some species occur in specific habitats such as tidal marshes (Mathieson et al., 2000; Akin et al., 2003), seagrass beds (Dorenbosch et al., 2006), mangrove forests (Barletta et al., 2000; Hindell and Jenkins, 2004), mangrove tidal creeks (Barletta et al., 2003; Krumme et al., 2005), and the main channel of the estuary (Barletta et al., 2005, 2008). Others species use these habitats and areas of the estuary (upper, middle, and lower) at various life stages and when environmental conditions allow (Thayer et al., 1987; Barletta et al., 2005, 2008; Dantas et al., 2010).

In tropical estuaries, seasonal fluctuations of salinity are a major factor determining larval abundance (Morais and Morais, 1994; Barletta-Bergan et al., 2002a; Barletta-Bergan et al., 2002b), and juvenile and adult biomass and density (Barletta et al., 2000, 2005, 2008; Dantas et al., 2010), mainly due to the effects of large fluctuations of freshwater inputs during the year (Figures 1, 2, and 3). In an estuary located at the eastern Amazon (northern Brazil), the estuarine-dependent species, which use the main channel, are ordered along a large-scale spatial gradient, when relatively stable hydrological conditions create a well-defined salinity gradient in the estuary during the late dry season (Barletta et al., 2005). Moreover, during the late rainy season, freshwater runoff increases, salinity drops, and the estuary then becomes more suitable for Neotropical freshwater species (Figure 1). Salinity and distance from the estuary mouth are the most important environmental variables structuring the fish assemblages in this estuary (Barletta et al., 2005). As an example, studies conducted in other Neotropical estuaries in South America (Barletta et al., 2005, 2008; Dantas et al., 2012) suggest that, for the marine catfish species *Cathorops spixii* (Agassiz) (Figure 4) and *C. agassizii* (Eigenmann and Eigenmann) (Figure 5), the salinity gradient influences the seasonal distribution not only of adults but also all of their different ontogenetic stages along the estuarine ecocline (upper, middle, and lower estuary). Moreover, the seasonal fluctuations in salinity (late dry and late rainy) define the nursery role for *C. spixii* and *C. agassizii* in the middle estuary (Barletta et al., 2005, 2008; Dantas et al., 2012) (Figures 4 and 5). The importance of this habitat as nursery for *C. spixii* and *C. agassizii* juveniles is determined by the seasonal environmental gradient conditions along the estuarine ecocline (Dantas et al., 2012).

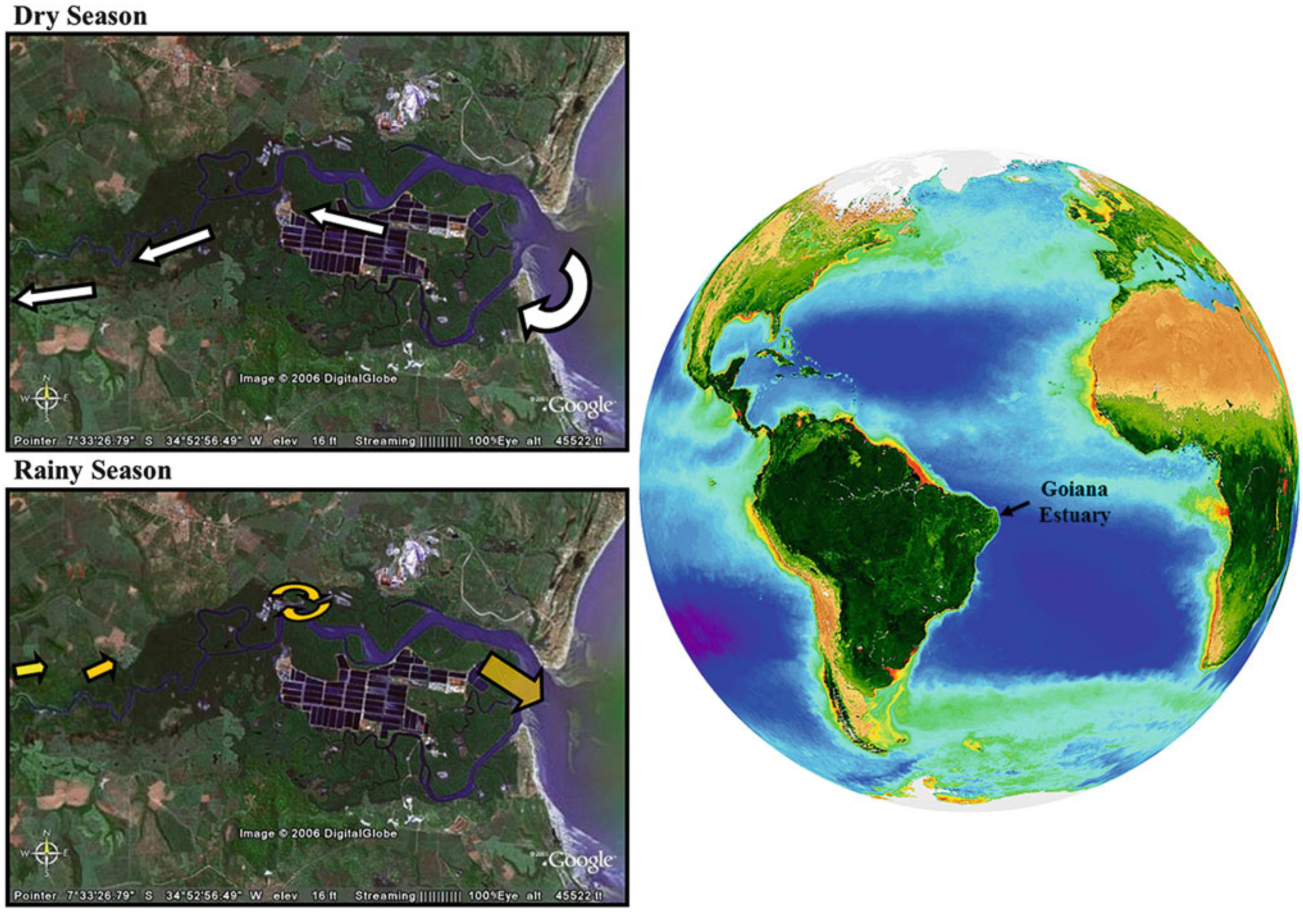
In the temperate La Plata River Estuary (Uruguay–Argentina), salinity has a stronger influence on the spatial structure of the fish assemblages than temperature, and the pattern of seasonal fish species distribution in the La Plata Estuary reflects the seasonal discharge of this river (Jaureguizar et al., 2004). On a large spatial scale, the strong physical environment gradient along La Plata Estuary creates a gradual change in the fish composition from fresh and shallow to marine and deeper waters that define the riverine, estuarine, and coastal shelf fish assemblages (Anganuzzi, 1983; Jaureguizar et al., 2004; Lorenzo Pereiro, 2007). The riverine fish assemblages occupy the inner part of the La Plata Estuary, characterized by



Fish Assemblages, Figure 1 Movement of fish assemblages induced by the seasonal fluctuations of salinity in the Caeté Estuary (eastern Amazon).

freshwater and shallow depths. The estuarine fish assemblage occupies the mixohaline waters of the La Plata Estuary, and its ichthyofauna is dominated by euryhaline species of marine origin. The species composition shows a gradient to the mouth, where estuarine resident species predominate and, to a lesser degree, occasional freshwater species, marine species, either stragglers or migrants, and estuarine migrants (Barletta et al., 2010).

In the intertidal mangrove creek habitats of an estuarine ecosystem in the northern Brazilian coast, fishes utilize the intertidal areas in different ways (Barletta et al., 2000, 2003; Barletta-Bergan et al., 2002b). Some fishes remain in the intertidal area at low tide, while others avoid this area during the low tide, using the habitat only when it is submerged. Two distinct fish assemblage patterns can be observed in the intertidal mangrove creeks in this region. Estuarine residents, marine seasonal migrants, and marine juvenile migrants represent the first fish assemblage, and marine juvenile migrants and marine adventitious visitors form the second fish assemblage. This pattern is strongly influenced by the seasonal fluctuation of abiotic parameters, mainly salinity. During the late rainy season, the freshwater runoff increases, salinity decreases, and juveniles of fresh-brackish-water adventitious visitors are found in the intertidal mangrove creeks during the flood tide (Barletta et al., 2003). Another strategy has been observed for fish species such as *Myrophis punctatus*, *Gobionellus smaragdus*, and *Kryptolebias* spp. They remain in the intertidal mangrove forest when it is not flooded (Figure 6). Barletta et al. (2000) identified three different strategies of use of this habitat during low tide by these species. The first group (G1) includes fish species that remain in crabholes until the next flood tide; the second group (G2) consists of fish species that remain buried or attached to *Rhizophora mangle* roots; and the third group (G3) is comprised of species which remain in the water streams (Figure 6). Independent of this fish assemblage variability, it is clear that seasonal variations in environmental gradients are the most important factor influencing the fish assemblage composition and structure in estuarine habitats (Barletta et al., 2010). In addition to the abiotic parameters (e.g., salinity, water temperature, dissolved oxygen, and turbidity) that influence fish assemblage characteristics, it is the combination of particular features of each estuarine system (e.g., climate, geology, geomorphology, latitude) that make them ever so valuable.



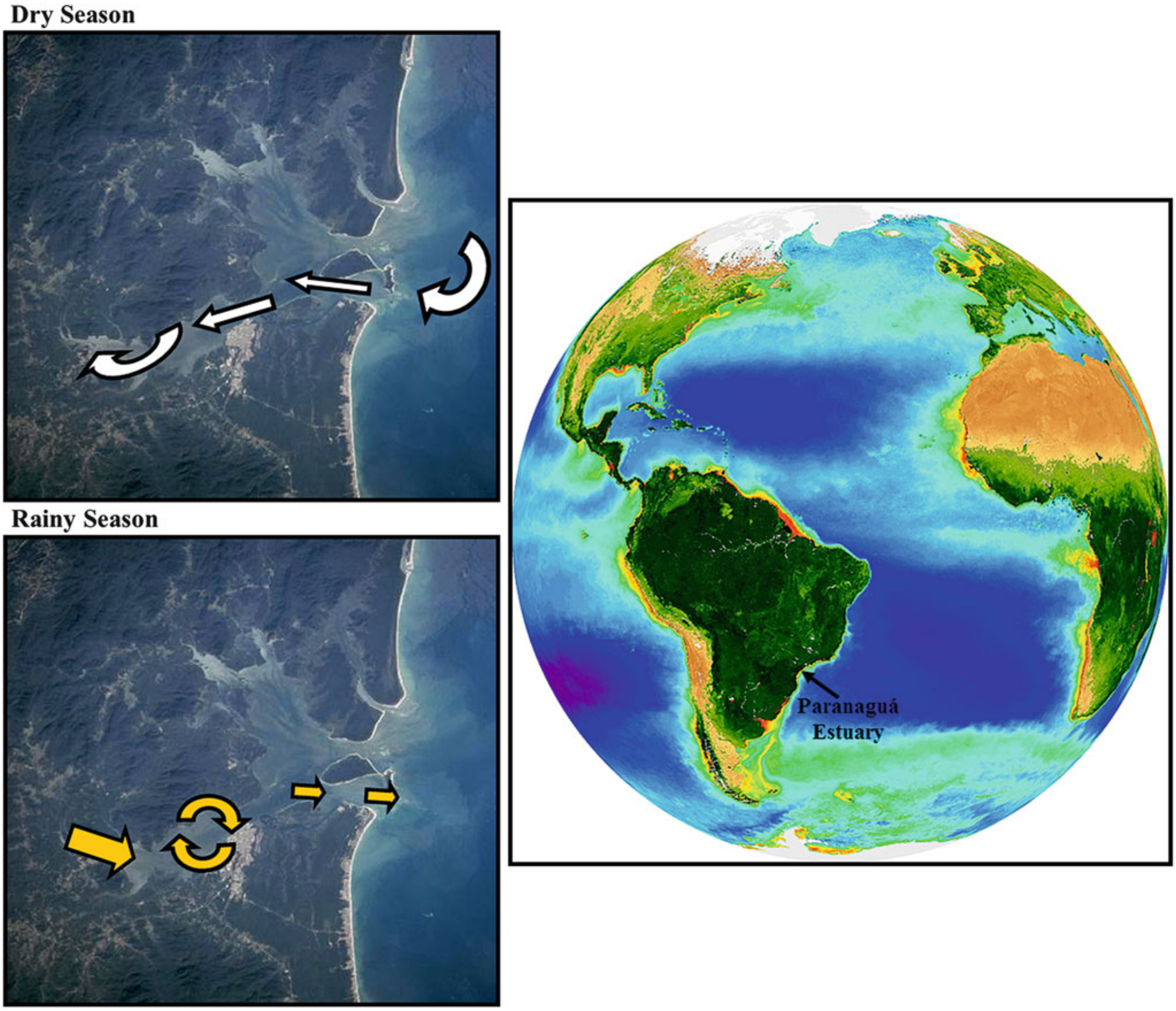
Fish Assemblages, Figure 2 Movement of fish assemblages induced by the seasonal fluctuations of salinity in Goiana Estuary (tropical semiarid of northeast Brazil).

Estuarine habitat roles

Estuaries in the tropical (Blaber, 2000; Barletta et al., 2005; Dantas et al., 2012), subtropical (Jaureguizar et al., 2004), and temperate regions (Thiel et al., 1995) act as breeding, mating, nursery, and feeding grounds for many fish species. Estuarine habitat use depends on a combination of the seasonal variation of abiotic factors (e.g., salinity, turbidity, temperature) and biotic factors (e.g., competition for food and space) (Blaber, 2000). For example, mangrove intertidal creeks (Barletta et al., 2000, 2003), main channels (Barletta et al., 2005, 2008; Dantas et al., 2012), and seagrass beds (Dorenbosch et al., 2006) provide different habitat for each fish species in a fish assemblage, depending on the ontogenetic phase of the species and the environmental conditions of specific sites during different times scales. In mangrove estuaries, the controlling marine influence is the tide, and up-estuary is the seasonal fluctuation of freshwater river flow (Barletta et al., 2010). Fish assemblages can be described by their biomass, density, and number of species, and these variables change in the estuary main channel, mudflats, and intertidal mangrove forest, at both high and low tides. The mangrove forest along the northern coast of South America is not flooded during low tide (tidal range, 5–7 m). Nevertheless, many fish species remain in the mangrove forest during this time. Tidal behavior strategies are described for fish species that linger in mangrove forests during low tide, using this habitat for protection from predation and for food resources (Barletta et al., 2000).

Estuaries support resident fish assemblages that are functionally important as an intermediate trophic level for many consumers. The production and seasonal occurrence of fishes vary with salinity, hydrology, and nutrient status in the estuary, and many species are adapted to these salinity fluctuations and are resident within estuarine habitats. Other species remain in estuaries only during certain periods of their life cycles and when the conditions are favorable (Blaber, 2000; Barletta et al., 2010).

Estuaries are frequently referred to as nursery areas for fishes (Beck et al., 2001). Ichthyoplankton can originate either from within the estuary or from adjacent marine and

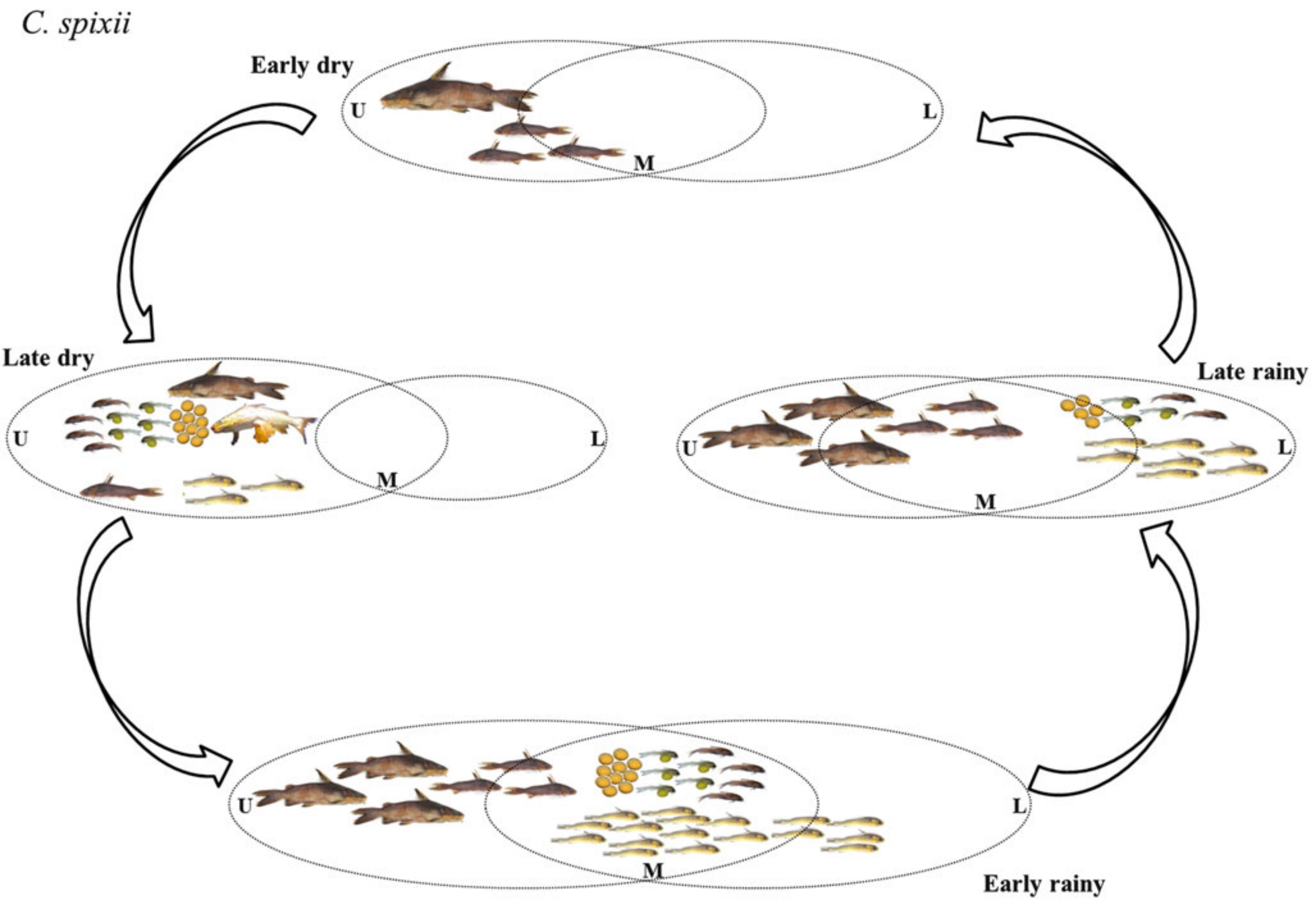


Fish Assemblages, Figure 3 Movement of fish assemblages induced by the seasonal fluctuations of salinity in Paranaguá Estuary (tropical to subtropical transition zone, south Brazil).

freshwater environments (Barletta-Bergan et al., 2002a). Juvenile fish use the different estuarine habitats as feeding grounds and refuge (Krumme et al., 2005; Dantas et al., 2013). The structure and seasonal dynamics of the fish larvae and juvenile fish assemblages clearly reflect the importance of the main channel (Barletta-Bergan et al., 2002a; Barletta et al., 2005) and mangrove forest (Barletta-Bergan et al., 2002b; Barletta et al., 2003) of the estuary as fish nursery habitats (Barletta and Blaber, 2007).

Estuarine connectivity

Estuaries are transitional zones between marine and freshwater systems, and the connection of these systems enables the transference of organisms, organic matter, and nutrients to nearshore coastal waters (Blaber, 2000; Barletta-Bergan et al., 2002a; Barletta-Bergan et al., 2002b; Barletta et al., 2003, 2005). Studies of estuarine fish assemblages show that they undergo large seasonal fluctuations in biomass and density, and estuarine-dependent species are ordered along a large-scale spatial gradient, when relatively stable hydrological conditions create a well-defined salinity gradient in the estuary (McLusky and Elliott, 2004; Barletta et al., 2005, 2008; Vilar et al., 2013). Understanding connectivity among estuarine habitats and populations of freshwater, estuarine, and marine fish is vital for studying population dynamics, managing fish stocks, designing marine



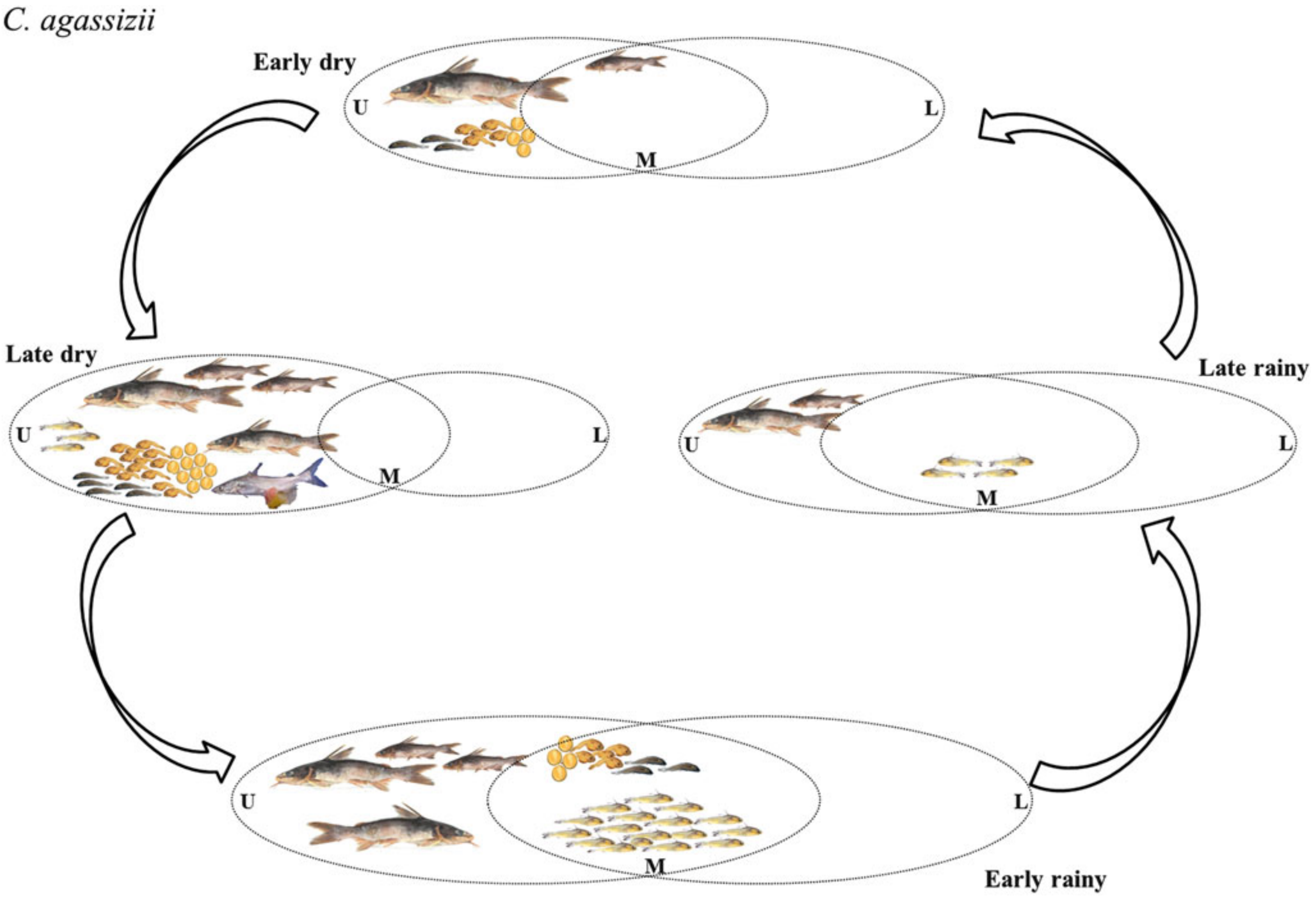
Fish Assemblages, Figure 4 Idealized model of the movement patterns of different ontogenetic phases of *Cathorops spixii* in the main channel (upper (*U*), middle (*M*), and lower (*L*)) of Neotropical estuaries along the western Atlantic coasts for each season (early dry, late dry, early rainy, and late rainy).

protected areas, and determining the patterns of habitat use by fish assemblages (Blaber et al., 2000; Gillanders, 2002; Barletta et al., 2010).

Many fish species use different habitats within estuaries, tidal marshes (Mathieson et al., 2000; Akin et al., 2003), seagrass beds (Dorenbosch et al., 2006), mangrove forests (Barletta et al., 2000; Hindell and Jenkins, 2004), mangrove tidal creeks (Barletta et al., 2003; Krumme et al., 2005), and the main channel of the estuary (Blaber, 2000; Barletta et al., 2005, 2008), only during a specific ontogenetic stage. These habitats are considered areas of feeding, mating, spawning, and nursery use for many fish species during a specific life stage, and the connectivity between these areas is very important to these species can complete their life cycles (McLusky and Elliott, 2004). The connectivity between habitats of an estuarine ecosystem is dictated not only by geomorphological factors but also by the environmental fluctuation of abiotic parameters, such as salinity, temperature, turbidity, and dissolved oxygen, which also influence the connection between the habitats and movement patterns of estuarine fish assemblages. Throughout the world, estuaries and adjacent coastal waters support numerous ecological and socioeconomic activities, but estuaries in particular are among the most modified and threatened of aquatic environments (Blaber et al., 2000). Human activities can drastically change the hydrological patterns of estuarine systems and can directly affect the connection between estuarine habitats, potentially impacting many ecological functions of these habitats for fish assemblages (Barletta et al., 2010).

Fisheries and estuarine habitats conservation

Coastal systems, including coastal lagoons (Mar Chiquita, Central Argentina; Rocha, Uruguay), estuaries (Caeté, northern Brazil; Paranaguá, south Brazil; Embley, Northern Australia; La Plata, Uruguay–Argentina), and bays (Samborombón Bay at La Plata Estuary, Anegada Bay in southern of Buenos Aires Province, Argentina), provide critical habitats for many commercial and recreational fish species (feeding, mating, spawning, and nursery grounds)

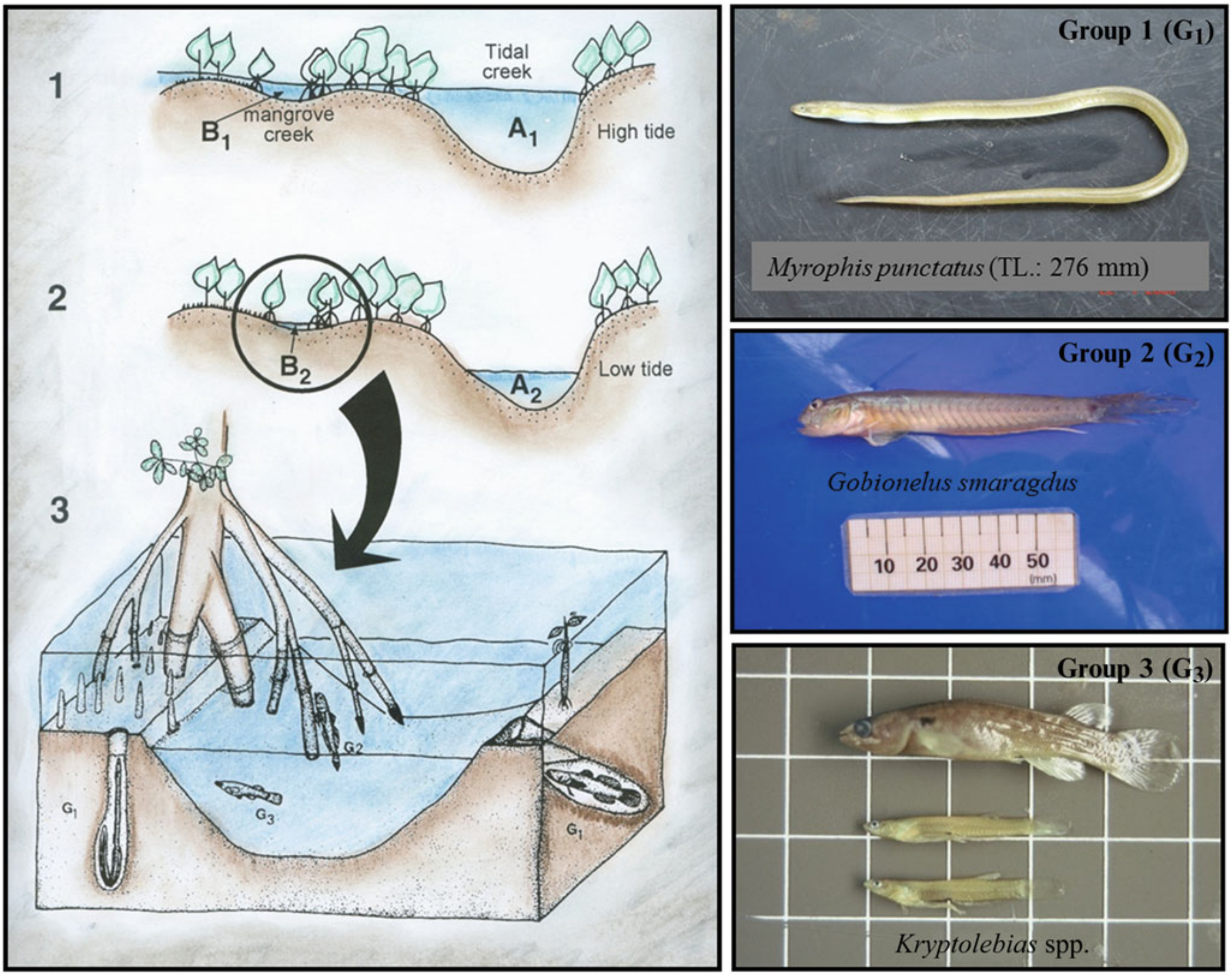


Fish Assemblages, Figure 5 Idealized model of the movement patterns of different ontogenetic phases of *Cathorops agassizii* in the main channel (upper (*U*), middle (*M*), and lower (*L*)) of Neotropical estuaries along the western Atlantic coasts for each season (early dry, late dry, early rainy, and late rainy).

and are characterized by wide variability of environmental conditions (Barletta et al., 2010). These systems also support human activities, such as fishing and environment use. Fisheries are an integral part of human societies in the coastal zone and are directly related and dependent on estuarine productivity (Blaber, 2000; Barletta et al., 2010). They can be divided into four main sectors: (1) subsistence (fishers consume their catch or give it away but do not sell it); (2) artisanal (fishers sell part of their catch but also retain part for their own consumption); (3) commercial (all catch is sold); and (4) recreational (fishing is carried out as a sport or leisure pastime and not primarily for producing food or income) (Blaber et al., 2000). Traditional fisheries have a long history and form part of the human culture in estuarine communities. They may also have a long-standing and complex interrelationship with the environment and in the tropics and subtropics are increasingly regarded as part of the overall ecology (Blaber et al., 2000). Implementing conservation strategies is essential in order to protect natural resources, including fishes and their habitats.

Monitoring and knowledge of the effects of small-scale and large-scale patterns in abiotic and biotic conditions, as well as fisheries activities, are necessary for a more complete understanding of fish and fisheries dynamics and, therefore, their effective management (Blaber et al., 2000; Barletta et al., 2010). Moreover, understanding the variations of fish assemblages in estuarine ecosystems at different spatial and temporal scales can provide valuable insights for fisheries management and nature conservation. The high spatial heterogeneity and the different fish fauna assemblages need a habitat-based classification of estuarine landscape features employing readily obtainable quantitative data for geophysical and oceanographic characteristics (Barletta et al., 2010). This categorization could provide data that correlate fish assemblages, or key species, to specific estuaries or coastal sectors. This would offer the necessary aid to regional fisheries management systems and provide a framework for research, monitoring, and conservation of the estuarine ecosystem.

Barletta et al. (2010) indicated that seasonal and annual variability in the southwest Atlantic coastal system



Fish Assemblages, Figure 6 Idealized cross sections through a typical intertidal mangrove forest in the coastal plain, indicating fish group distribution strategies during low tide. *A1* and *A2* represent the main tidal channel; *B1* and *B2* represent the very small creeks in the intertidal mangrove forest. Cross section 1: intertidal mangrove forest during high tide; cross section 2: intertidal mangrove forest during low tide; cross section 3: intertidal mangrove creek during low tide (*B2* situation – vertical scale exaggerated). Cross section 3 represents the three different ecological strategies developed by fish to reduce interspecific competition in the intertidal mangrove forest during low tide. The first strategy (Group 1 – *G1*) includes fish species that stay in crabholes until the next flood tide. Group two (*G2*) represents the fish species that stay buried or attached to *Rhizophora mangle* roots. The third group (*G3*) represents the species that stay in the water stream (Modified from Barletta et al., 2000).

influences various fish biological behaviors (migration, spawning, mating, and bottom fidelity), as well as the seasonal and interannual coastal habitat use, that determine fish catchability and susceptibility. On a large spatial scale, the strong physical environmental gradient along the southwest Atlantic coastal system creates a gradual change in the fish composition from fresh and shallow to marine and deeper waters that define the riverine, estuarine, and coastal shelf fish assemblages (Anganuzzi, 1983; Jaureguizar et al., 2004; Lorenzo Pereiro, 2007). The riverine fish assemblages occupy the inner part of the La Plata River characterized by freshwater (salinity 0.08 ± 0.02) and shallow depths (7.88 ± 1.53 m). Its ichthyofauna forms part of the Paranoplatense fish community and shows a high affinity for the Paraná and Uruguay River basins (Cousseau, 1985). It is mainly dominated by freshwater and anadromous species. The anadromous species, during spring and early summer, extends up the La Plata River and its basin to spawn. The estuarine fish assemblage occupies the mixohaline waters of the La Plata Estuary, and its ichthyofauna is dominated by euryhaline species of marine origin. The species composition shows a gradient to the mouth where estuarine resident species predominate and, to a lesser degree, occasional freshwater species, marine species, either stragglers or migrants, and estuarine migrants. The fish assemblages can be described in relation to biomass, density, and number of species and the seasonal fluctuations of environmental variables (salinity, water temperature, and dissolved oxygen).

In tropical estuaries from the western Atlantic Ocean, these variables change in the estuarine main channel and intertidal mangrove forest at both high and low tides (Barletta et al., 2010). The estuary then supports a resident fish community that is functionally important as an intermediate trophic level for many consumers. The production and seasonal occurrence of fishes appear to vary with salinity, hydrology, and nutrient status in the estuary, all of which are controlled by both freshwater flow and tide. Many species are adapted to these salinity fluctuations and are resident within the estuarine habitats. Others stay in the estuary only during a certain period of their life and when conditions are appropriate. Estuaries are frequently referred to as nursery areas for both fishes and invertebrates (Beck et al., 2001).

Intertidal mangrove areas are also considered essential nursery grounds for fish fauna and often include commercially important species (Morton, 1990; Laegdsgaard and Johnson, 1995; Barletta et al., 2003). The northern Brazilian coast (tropical and humid) houses the largest continuous mangrove estuarine belt in the world and constitutes 56.6 % of the mangroves in South America (7,592 km^2) (Souza Filho, 2005). South of the mouth of the Amazon River, more than 30 estuaries fringe ~650 km of mangrove-dominated coastline, and many fish species use intertidal creeks for tidal migrations between mangrove habitats and adjacent subtidal areas, the main channel, and coastal waters (Barletta et al., 2010). This connection between mangrove and adjacent estuarine coastal habitats is an important function for fish and fisheries in coastal waters, providing nutrient support and establishment of ecological functions for estuarine fish assemblages. In order to protect natural resources, including fishes and fish habitats, decision makers need to plan and implement national/regional conservation programs, including the formation of conservation units (CU).

Summary

A fish assemblage is simply a suite of species whose individuals are collected in the same area and at the same time. There are no strict limits on the spatial or temporal scale of a fish assemblage (Miller, 2002). It is correct to speak of the assemblage in a single season and region or at specific times and locations. Research on fish assemblages has sought to understand why they occur in the places and at the times they do and how these spatial and temporal distributions relate to patterns in breeding, nursery area, recruitment, and foraging. Understanding the roles and relative importance to fish of different habitats within estuaries is necessary for both effective management and conservation (Elliott and Hemingway, 2002). The information on estuarine fish community structure and functioning is important to understanding the ecological services of estuaries and is also important to classify and categorize estuarine fauna as an aid to understanding and managing the effects of human activities on estuaries (McLusky and Elliott, 2004; Elliott et al., 2007). Traditional fisheries have a long history and comprise part of the human culture of estuarine communities. They may also have a long-standing and complex interrelationship with the environment and in the tropics and subtropics have become part of the overall ecology (Blaber et al., 2000). Implementing conservation strategies is essential to protect natural resources, including fishes and fish habitats.

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Cross-references

Anadromous
Ichthyofauna
Ichthyoplankton

FJORD

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Synonyms

Fiord

Definition

A fjord is a sea inlet that has inundated a trough formed by glacial erosion. Like the glaciers that formed the troughs, fjords range in length from a few to hundreds of kilometers and are common in the modern and past glaciated portions of the Arctic, Antarctic, South America, northwest Europe and Scandinavia, northern North America, and New Zealand.

Origin and morphology

Although troughs erode rapidly under glacial conditions, many fjords are erosionally complex, palimpsest features that formed over the course of multiple glacial episodes (Nesje and Whillans, 1994; Bennett and Glasser, 2009). Because glacial erosion is dependent on ice thickness, erosion rates are highest on the valley floors, resulting in the common steep-sided U-shaped valley forms. These cross-sectional profiles can be described mathematically by empirical power-law functions or second-order polynomials (Harbor and Wheeler, 1992; Harbor, 1995; James, 1996; Amerson et al., 2008). However, profile asymmetry is common due to both the physical characteristics of the underlying bedrock and subsequent erosional and depositional processes after inundation (Augustinus, 1992; Nesje and Whillans, 1994; Augustinus, 1995). The longitudinal profile of a fjord depicts the erosional and depositional characteristics of the formative glacier system with the deepest erosion and subsequent inundated water depths, coinciding with the area of maximum ice discharge (Yingkui et al., 2001). Reduced bedrock erosion and/or moraine deposition near the terminus of the past glacier extent results in a shallow sill near the mouth of the fjord. The sills and overdeepened troughs allow fjords to accumulate a unique sedimentary archive of past marine and glacial conditions (Benn and Evans, 2010). This same bathymetric juxtaposition also causes extreme currents and inhibits exchange of bottom waters between the fjord and adjacent sea (Farmer and Huppert, 1979; Farmer and Freeland, 1983). The seasonal and annual mixing of terrestrial and oceanic waters, coupled with various sea ice and glacier inputs at high latitudes, yields complex temperature, salinity, and density gradients both vertically and between the head and the mouth of the fjord (Farmer and Huppert, 1979; Farmer and Freeland, 1983).

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Cross-references

Coastal Bays
Firth

FLOCCULATION

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Synonyms

Aggregation; Coagulation

Definition

Flocculation is a process of contact and adhesion whereby dispersed particles are held together by weak physical interactions.

Introduction

Flocculation has been widely used in water and wastewater treatment applications mainly for clarification and reduction of suspended solids, respectively. Recently, it has also been used as a cost-effective method for harvesting microalgae for biomass for production of food, feed, fuel, or chemicals (Vandamme et al., 2013). Aside from these applications, in situ flocculation has also been studied for years to understand sediment transport in different aquatic environments. In this regard, Eisma (1986) defined flocculation as a natural process by which suspended particles are brought together into larger units called flocs.

Flocculation or floc formation depends on the physical collision between suspended matter and their adhesion (Alldredge and Jackson, 1995; Hansen et al., 1995). Droppo (2001) described flocs as heterogeneous, composite structures composed of an active biological component, a nonviable biological component, inorganic particles, and water held within or flowing through pores. Because of the diverse origins of flocs, their characteristics are highly variable (Alldredge and Silver, 1988). These can also be viewed as individual microecosystems with autonomous and interactive physical, chemical, and biological functions or behaviors operating within the floc matrix (Droppo et al., 1997). Microflocs are dense, quasi-spherical, resistant to turbulent mixing, and small with sizes ranging from 100 to 160 μm (Verney et al., 2009). Under favorable conditions, microflocs collide with each other, flocculate, and form macroflocs (Simon et al., 2002; Verney et al., 2009). Macroflocs are formed from microflocs up to several millimeters and can rapidly disintegrate back into microflocs.

Factors affecting flocculation

Flocs are formed within the water column or on the surface of a bed by a variety of complicated physical, physicochemical, and biological means (Droppo, 2001; Simon et al., 2002). The physical mechanisms that bring particles together in the ocean are Brownian motion, fluid shear, and differential settlement (Mccave, 1984; Alldredge and Silver, 1988; Simon et al., 2002). Brownian motion dominates interactions of fine particles (less than 8 μm). Differential settlement should dominate coagulation between similarly sized particles between 1 and 100 μm in surface waters. It is important for sinking of particles in the water column and in slack water in tidally affected shallow seas and estuaries (Simon et al., 2002). McCave's (1984) calculations indicate that collisions of small particles with larger ones resulting in floc formation should be controlled primarily by shear. Shear collides similarly sized particles and leads to scavenging of small particles by large ones more effectively than differential settling. In agreement, a study conducted by Chen et al., (2005) in Scheldt reported that floc formation in the estuary was rather controlled by current velocity and the suspended matter concentration than salinity.

Flocs are composed of biological components, inorganic particles, and water that carry with them negative *surface charge*, hence, are affected by varying pH and salinity. Salinity is used by oceanographers as a measure of the total salt content of seawater. Clay particles which are usually negatively charged have a high cationic adsorption capacity. Interparticle forces then become attractive at increased salinities, causing particles to collide and flocculate (Day et al., 1989). pH is a measure of acidity or alkalinity which can either increase (e.g., salt influences) or decrease (organic matter load) due to the ionic composition of the system. It has been reported in some experiments that changes in surface charge due to pH variations affect floc stability (Wilén et al., 2000).

Another factor called *bridging* occurs when the loops and tails of a polymer adsorbed to one particle become attached to one or more other particles (Droppo et al., 2005). Divalent cations such as Ca^{2+} and Mg^{2+} can in some cases act as chemical bridging agents between negative charges of the polymers and negative charges on the particle surface, enhancing the attraction and sticking properties of electronegatively charged particles and polymers (Simon et al., 2002).

Flocculation can be biologically mediated by either particulate or dissolved *organic matter*. Organic matter is adsorbed at the surface of sediment particles, giving it a negative charge. Sticky organic compounds (transparent exopolymer particles) produced in the water column by phytoplankton, bacteria, and macrophytes also promote aggregation and sedimentation of particles (Passow, 2002).

Summary and conclusion

Diverse studies on flocculation have been conducted through the years. These studies lead to better understanding of the factors affecting flocculation at given environmental conditions. Moreover, applications of flocculation studies can be valuable to society such as in water and wastewater treatment, management of harmful algal blooms, and siltation control.

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Cross-references

pH
Sediment Transport
Tides

FLUSHING TIME

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Synonyms

e-Folding time; Freshwater replacement time; Freshwater residence time, Freshwater transit time; Freshwater turnover time

Definition

The flushing time of an estuary is generally defined as the turnover time of freshwater in the estuary (τ_{fw}), that is, the time required to replace the freshwater contained in the estuary with freshwater inflow. The flushing time of an estuary is calculated as the ratio of the volume of freshwater in the estuary (V_{fw}) to the total rate of freshwater input (Q_{fw}):

$$\tau_{fw} = \frac{V_{fw}}{Q_{fw}}$$

While V_{fw} increases as Q_{fw} does, it does so more slowly, so that flushing time decreases as freshwater flow increases (Pilson, 1985).

Measurement

The freshwater content (V_{fw}), and therefore τ_{fw}, may be determined by mass balance calculations from the estuary volume, the volume-weighted average salinity in the estuary, and the salinity outside the seaward boundary (Pilson, 1985). This is termed the freshwater replacement method.

Flushing time may also be estimated by introducing a conservative tracer, such as dye, at a constant concentration into the freshwater inflow until the mass or average concentration of tracer in the estuary at a given tide stage attains equilibrium. After termination of tracer input, the spatially averaged concentrations will decrease approximately exponentially as tracer is flushed from the estuary. The time required for the concentration to attain e^{-1} times the initial concentration, often referred to as the e-folding time, is an estimator of the flushing time.

Applications

Flushing time is a useful indicator of the behavior of materials introduced into an estuary with freshwater. For instance, the fraction of nitrogen entering the estuary from the watershed that flows through the estuary to the sea, and the fraction lost within the estuary to processes such as denitrification and permanent burial in sediments, may be estimated using the flushing time (Dettmann, 2001). The turnover or mean transit time of conservative materials introduced with freshwater are equal to the flushing time of freshwater, while those of nonconservative materials that are consumed by processes in the estuary have shorter turnover times. See Dettmann (2008) for details in an application to freshwater lakes.

Flushing time, as described above, applies to an estuary as a whole. The concept of a flushing time may also be applied to a portion of an estuary, e.g., in a box model (Hagy et al., 2000).

Other concepts related to flushing time are sometimes used in describing material movement through an estuary. Examples are estuarine residence time, that is, the residence time in the estuary of a conservative substance introduced uniformly in concentration throughout the estuary, and pulse residence time (the residence time of a conservative substance introduced as an instantaneous pulse in a limited portion of the estuary) (Hagy et al., 2000).

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FOOD CHAIN

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Synonyms

Trophic levels

Definition

Food chain refers to the transfer of energy (linear sequence of links) from the base to the top of a food web. The number of trophic levels is often called the food-chain length.

Description

Food-chain length is a central characteristic of estuarine ecosystems. It reflects the organization of trophic interactions in biotic communities (Pace et al., 1999; Persson, 1999; Oksanen and Oksanen, 2000); key ecosystem functions, such as nutrient cycling, primary productivity, and atmospheric carbon (C) exchange (Pace et al., 1999); and the bioconcentration of contaminants in top predators, including many fish that humans consume (Spencer and Warren, 1996; Kidd et al., 1998). Conventional wisdom holds that food-chain length is determined either by the dynamic stability of food webs or by the availability of limiting food resources (often represented as energy). Some recent studies strongly challenge this conventional wisdom (Post et al., 2000), whereas others reframe the question to accommodate functional definitions of food-chain length (Oksanen et al., 1996). These and other studies suggest a complex relationship between food-chain length and ecological processes, such as the history of community organization, resource availability, habitat stability, and ecosystem size.

Different hypotheses predict food-chain length to be determined by productivity alone (productivity hypothesis) (Diehl and Feissel, 2001), ecosystem size alone (ecosystem-size hypothesis) (Kitching, 2000], or a combination of productivity and ecosystem size (productive-space hypothesis). The productivity and productive-space hypotheses propose that food-chain length should increase with increasing resource availability; however, the productivity hypothesis does not include ecosystem size as a determinant of resource availability. The ecosystem-size hypothesis is based on the relationship between ecosystem size and species diversity, habitat availability, and habitat heterogeneity (Menge and Sutherland, 1987).

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Cross-references

Detritus Food Webs
Food Web/Trophic Dynamics

FOOD WEB/TROPHIC DYNAMICS

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Synonyms

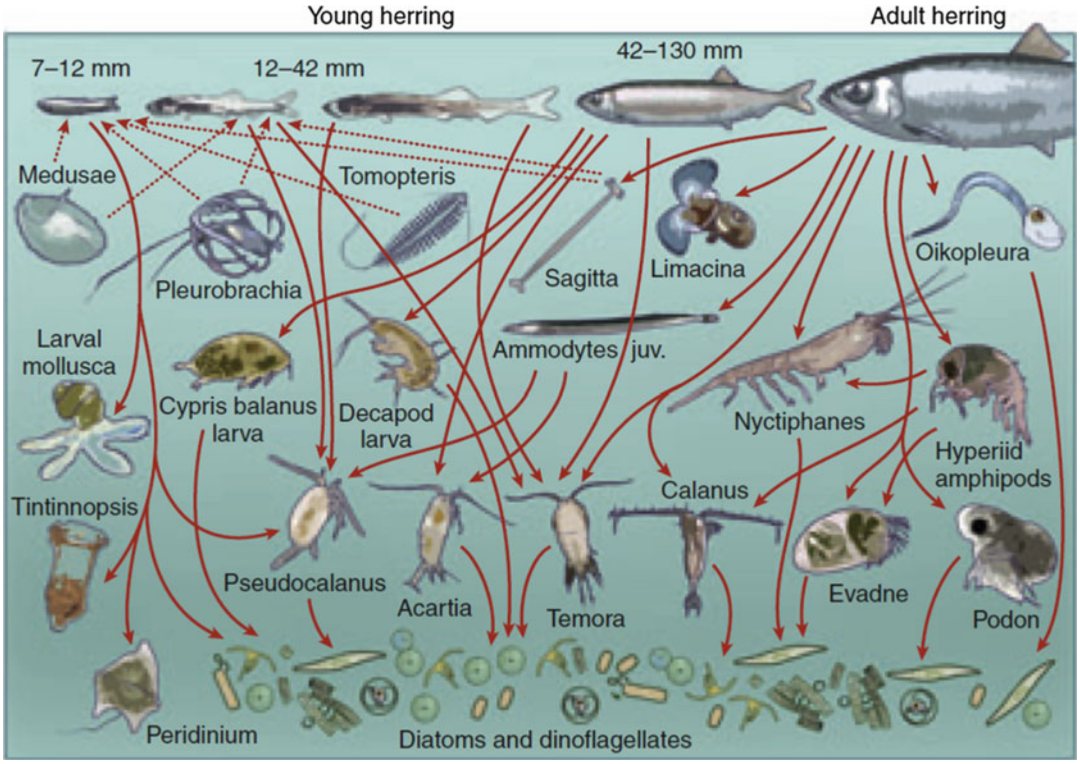
Food cycle; Food nexe; Trophic network

Definition

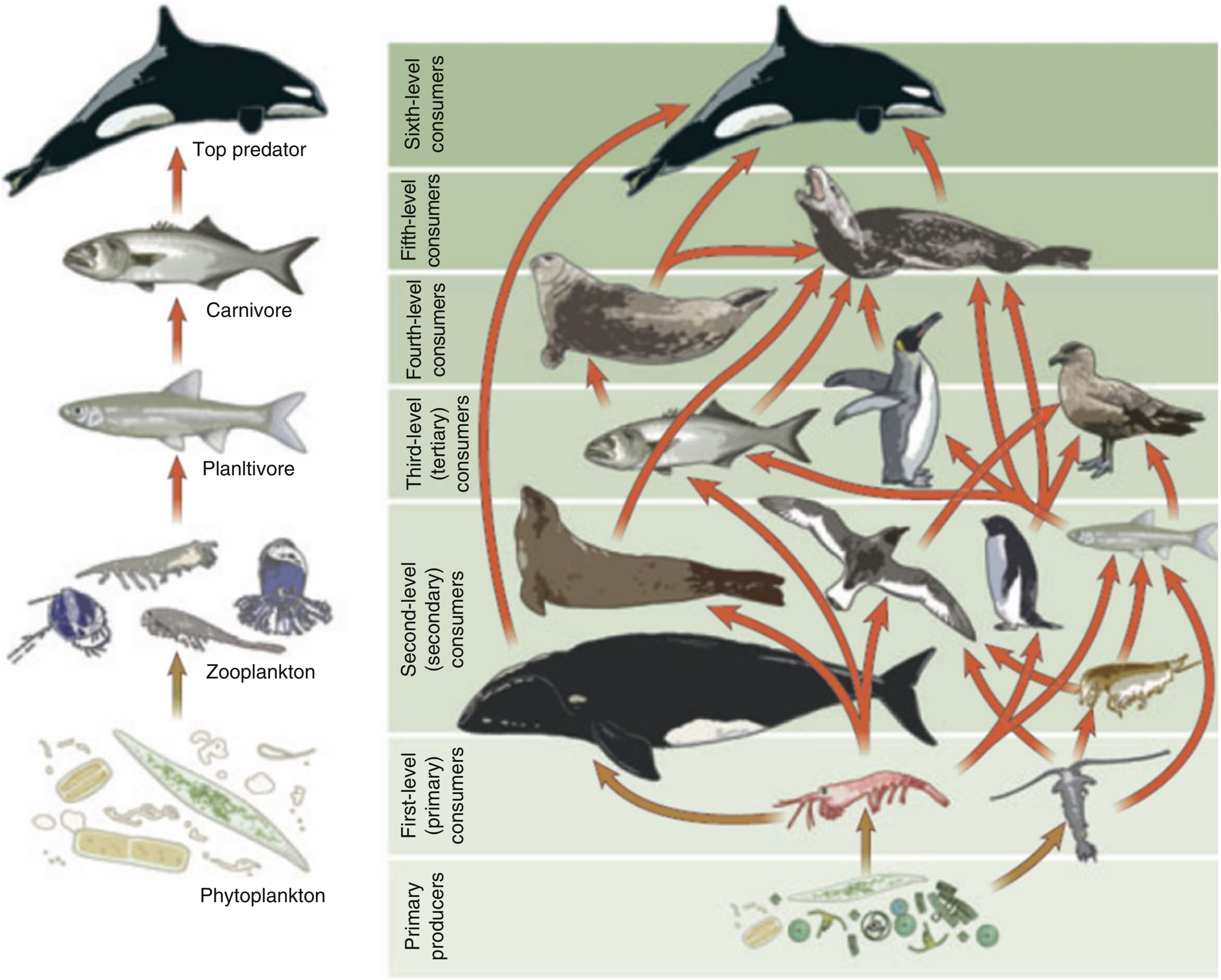
The network of interconnected food chains in an ecosystem, wherein organisms that eat each other form a sequence of interconnecting links from the initial producers of organic matter through the various consumers.

Description

Believed to have been initially depicted by the Muslim scientist al-Jāḥiẓ (nickname, meaning "google-eyed" for Abu 'Uthman' Amr ibn Bahr) in the ninth century (Egerton, 2002), food chains communicate the procession of energy from the origin of fixation by primary producers through herbivores and successive levels of higher level consumers. Early formulations by Sir Alistair Hardy (1924, Figure 1) and Charles Elton's *Animal Ecology* (1927 and subsequent volumes) amalgamated food chains into food webs (Figure 2). Transfer of energy through the respective trophic levels from one part of an ecosystem to another was further articulated by Lindeman (1942) in his transformative ideas of trophic-dynamic relationships to the process of ecological succession. Winemiller and Polis (1996) stated that, "With few exceptions (fossilization, mineralization), the ultimate fate of organisms is some



Food Web/Trophic Dynamics, Figure 1 The classical food web, based on Hardy (1924). Page 294 from Chamberlin, W.S., and T.D. Dickey. 2008. Exploring the Ocean World. McGraw-Hill Higher Education; ISBN: 0073016543.

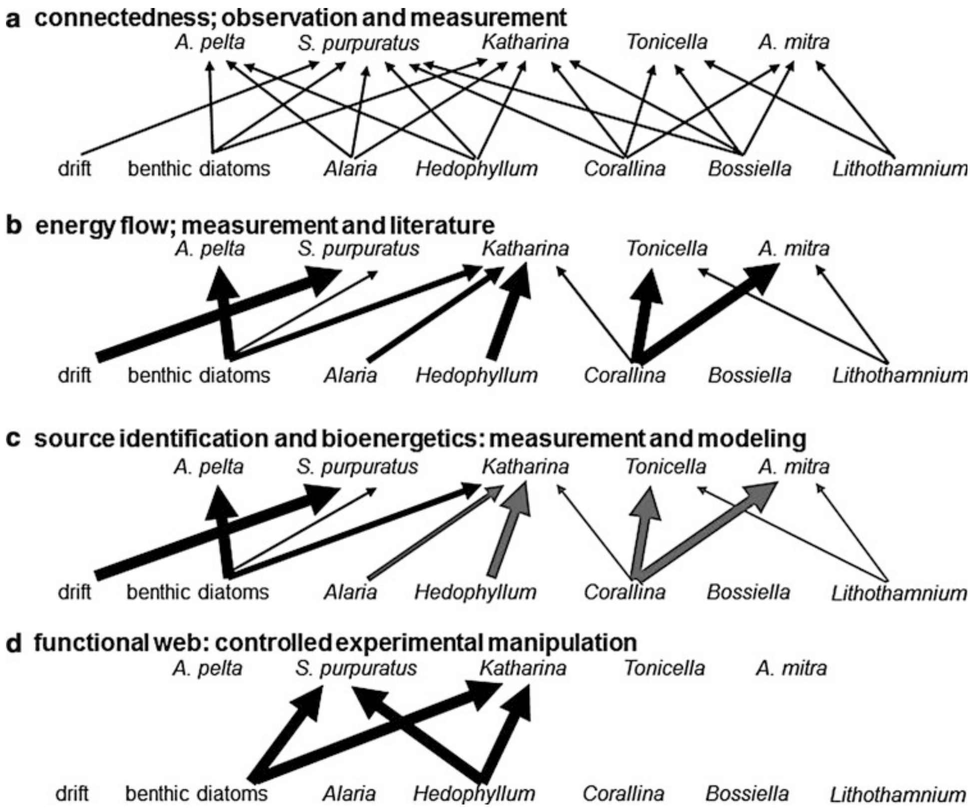


Marine food chain

Marine food web

Food Web/Trophic Dynamics, Figure 2 Food chains versus food webs. (**a**) The linear transfer of energy and matter can be depicted in a simple food chain. In the open ocean, these chains become quite long. (**b**) Food webs included all of the possible pathways of exchange of energy and materials among organisms. Food webs are especially useful where organisms feed at multiple trophic levels during different stages of their life. Page 290 from Chamberlin, W.S., and T.D. Dickey. 2008. Exploring the Ocean World. McGraw-Hill Higher Education; ISBN: 0073016543.

form of consumption by and assimilation into tissues of other organisms, be they metazoans or microbes. The structure, dynamics, and spatial relationships of the trophic networks derived from this basic observation are certain to affect the distribution and abundance of organisms in very fundamental ways." The dynamics of food webs are now viewed from multiple perspectives. Paine (1980) effectively articulated three synthetic characterizations of both the structure and dynamics of food web links affecting community structure, from (1) static, topological connectedness to (2) energy flow and (3) the strength of functional interactions among the connected species (Figure 3). With revelations of contaminant biomagnification (Suedel et al., 1994), the role of the "microbial loop" (Azam et al., 1983), emergence of intrinsic patterns (Pimm, 1982), and the use of elaborate ecological models (e.g., Ecopath, Ecosim and Ecospace; Pauly et al., 2000) and geochemical biomarkers such as natural stable isotopes and fatty acids (e.g., Hanson et al., 2010) have greatly expanded our delineation and application of



Food Web/Trophic Dynamics, Figure 3 Four conceptual and empirical approaches to characterizing trophic relationships (Modified from Paine's (1080; Food webs: linkage, interaction strength and community infrastructure; Journal of Animal Ecology 49: 666–685; British Ecological Society, Blackwell Publishing LTD, John Wiley & Sons; ISBN: 0021-87980) example from rocky intertidal community. The connectedness web (**a**) is based on observation, the energy flow web (**b**) on measurements and literature values, the stable biomarker and bioenergetic web on measurements and modeling, and the functional web (**d**) on controlled experimental manipulation. The size of *arrows* in (b) through (**d**) indicate strength of energy transfer or community interaction; the shaded arrows in (**c**) implies less source transfer or net bioenergetic benefit to consumer.

complex food webs. An understanding of the cascading effects of functional interactions among species across communities and ecosystems remains at the core of trophic dynamics.

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Cross-references

Detritus Food Webs
Food Chain

FOREDUNE

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Synonyms

Beach ridge; Frontal dune; Primary dune; Transverse dune

Definition

A foredune is a shore-parallel dune ridge that forms on the back beach of an ocean, estuary, or bay by aeolian depositional processes (Hesp, 1988; Nordstrom, 1992; Nordstrom and Jackson, 1994).

Description

As part of a dune system, the foredune is located closest to the water body. Vegetation in the backshore plays an important role in the accumulation and retention of sand to form the dune ridge. Plant succession occurs as the dune develops. Foredunes often exhibit different morphologies (from convex ridges to flat terraces) and variable ecological characteristics (Hesp, 1988). As the foredune enlarges, it traps more windblown sand and hence can expand rather rapidly.

There are two main types of foredunes: incipient and established types (Hesp, 2002). Incipient foredunes are new or embryo dunes formed by sand deposition within discrete vegetation (e.g., *Ammophila*, *Ipomoea*, and *Spinifex*) from the immediate backshore to back-barrier flats (Carter et al., 1992). They develop into established foredunes (up to 30–35 m in height) with the growth of larger and more mature vegetation, notably woody plants. A number of factors control the developmental process, including the plant species present, degree of vegetation cover, sand supply, accretion rate, wave and wind forces, storm erosion, extent of human impact and use, as well as other elements (Hesp, 2002).

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Cross-references

Back Dune
Beach Processes
Coastal Barriers
Coastal Landforms
Estuarine Beaches

FORESTED WETLAND HABITAT

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Synonyms

Mangrove forest; Swamp; Tidal freshwater forested wetland; Tidal saltwater forested wetland; Tidal várzea

Definition

A forested wetland (swamp) is a forest where soils are saturated or flooded for at least a portion of the growing season, and vegetation, dominated by trees, is adapted to tolerate flooded conditions.

A tidal freshwater forested wetland is a forested wetland that experiences frequent but short-term surface flooding via tidal action, with average salinity of soil porewater less than 0.5 g/l. It is known locally as tidal várzea in the Amazon delta, Brazil.

A tidal saltwater forested wetland (mangrove forest) is a forested wetland that experiences frequent but short-term surface flooding via tidal action, with average salinity often exceeding 3 g/l and reaching levels that can exceed seawater. Mangrove ecosystems are composed of facultative halophytes that generally experience better growth at moderate salinity concentrations.

Introduction

Forested wetlands represent a globally diverse array of types but are restricted to only a few true estuarine types. The primary component differentiating various types of forested wetland habitat is the vegetation, which is dominated by trees and shrubs. The specific composition of tree and shrub species (i.e., the community) is mostly influenced by the local hydrology (Mitsch and Gosselink, 2000). Because estuarine systems are dominated by the oceanic processes of sea level and tidal action on one side and the continental processes of river discharge and sediment load on the other, hydrologic characteristics within the estuary vary considerably. Depth, duration, and frequency of flooding; salinity of the soil porewater; and regional climate all determine the type of forested wetland habitat. Estuarine forested wetlands are divided into tidal freshwater forested wetlands and mangroves, also known as tidal saltwater forested wetlands.

Tidal freshwater forested wetlands

These wetlands occur within floodplains of coastal rivers at the upper boundary of tidal influence, often just upstream of tidal marshes. Larger areal distributions of tidal freshwater forested wetlands are generally found in areas that have large tidal ranges coupled with high river discharge (Conner et al., 2007). However, they are not restricted to these conditions and can occur in smaller watersheds. Conditions are favorable for this habitat type worldwide, but most published accounts describe conditions in the southeastern United States. Studies outside the United States have been limited to Central America and the Amazonian coast in South America, with limited descriptions (see Verhoeven et al., 2001; Struyf et al., 2009) of willow (*Salix* spp.)-dominated tidal freshwater forested wetlands in Europe.

A frequently described characteristic of these wetlands includes the prominent development of microtopographic patterning under purely freshwater conditions. This patterning is often called hummock and hollow topography (Rheinhardt and Hershner, 1992) and is common in these wetlands when occurring in non-stressed states (Conner et al., 2007). Hollows are low-lying, flat areas that are mostly bare mud or contain herbaceous vegetation similar to tidal freshwater marshes. Hummocks are raised microsites that average 15–20 cm high and are roughly 1–10 m^2 in size. Hollows are flooded during most flood tides and remain saturated within 20 cm of the surface nearly 100 % of the time. In contrast, hummocks typically flood less frequently, to lesser depths, and remain saturated for shorter times, which increases oxygen penetration and affects nutrient availability (Courtwright and Findlay, 2011). As a result, hummocks often contain a greater diversity of tree and shrub species, especially in remote backswamp areas (Duberstein and Conner, 2009).

Hydrology and community composition

Rivers flowing toward the ocean are impeded during flooding tides, first resulting in flow reversals within the channel and then rising water levels, often resulting in overbank flooding onto the floodplain. The frequency, depth, and duration of flooding are determined primarily by lunar- and wind-driven tides (Conner et al., 2007), and the salinity of the floodwater ranges from full-strength seawater (35 g/l) to completely fresh (<0.5 g/l) depending upon the relative contributions of seawater versus fresh river water over multiple tidal cycles. However, floodwater over the soil surface in tidal freshwater forested wetlands typically has low salinity, thus keeping the soil porewater fresh or nearly so. Storm surges can bring pulses of saline water into tidal freshwater forested wetlands, but high salinity floodwater usually leaves the floodplain relatively quickly, allowing for freshwater flushing. Tidal freshwater forested wetlands have, by definition, average annual soil porewater salinities <0.5 g/l; however, sometimes, salinity pulses, e.g., from storm surge or drought, are incorporated into the porewater, and when that happens, habitat change to oligohaline (low salinity, 0.5–5.0 g/l) marsh can occur (Brinson et al., 1985).

Tidal freshwater forested wetlands within the southeastern United States vary more in the relative dominance and density of tree species, rather than presence or absence, with some exceptions. The most common canopy trees include swamp tupelo (*Nyssa biflora*), water tupelo (*Nyssa aquatica*), baldcypress (*Taxodium distichum*), pumpkin ash (*Fraxinus profunda*), Carolina ash (*Fraxinus caroliniana*), green ash (*Fraxinus pennsylvanica*), and red maple (*Acer rubrum*). Shrub species tend to vary more between river systems, but the most ubiquitous are hazel alder (*Alnus serrulata*) and wax myrtle (*Morella cerifera*). Atlantic white cedar (*Chamaecyparis thyoides*) is rare in tidal freshwater areas but can be found in isolated stands in the southeastern United States. It occurs in fairly monotypic stands along the coast of North Carolina or as part of a diverse mix of hardwood species restricted to parts of Mississippi near the Alabama border (Conner et al., 2007). Swamps in Louisiana contain primarily baldcypress and water tupelo, but in other parts of the southeastern United States, the diversity of trees and specific assemblages follow a salinity and flood frequency gradient (Conner et al., 2007) with the most frequently flooded and most saline stands consisting primarily of baldcypress in the canopy with wax myrtle in the understory (Krauss et al., 2009).

Central American and Amazonian tidal freshwater forested wetlands are often managed for agriculture (e.g., cacao, assai), though some unmanaged landscapes still exist. There are two general types of unmanaged tidal freshwater forested wetlands in Central and South America: palm swamps and hardwood swamps. Tidal palm swamps generally occur as low-diversity patches within hardwood swamps, with virtual monocultures of the dominant tree or palm species (Prance, 1979). Hardwood swamps in Central America also have very low diversity (Ellison, 2004). Amazonian tidal várzea are dominated by relatively few species as well but have higher total diversity than any other tidal freshwater forested wetland reported thus far in the scientific literature when left unmanaged (see Almeida et al., 2004). Tidal palm swamps in Honduras are dominated by the spiny palm (*Bactris minor*), whereas yolillo palm (*Raphia taedigera*) dominates in Costa Rica and Amazonia. Amazon palm swamps can also be dominated by muriti (*Mauritia flexuosa*), assai (*Euterpe oleracea*), or troolie (*Manicaria saccifera*). Hardwood tidal swamps in Honduras are dominated by dragonsblood tree (*Pterocarpus officinalis*) with coin vine (*Dalbergia ecastophyllum*, a shrub) common in the understory. Tree species in tidal várzea vary widely between stands and subregions, but perhaps the most common include baboonwood (*Virola surinamensis*), tornillo (*Cedrelinga catenaeformis*), silk cotton tree (*Ceiba pentandra*), and pracuiba (*Mora paraensis*) (Prance, 1979). Much like tidal freshwater swamps in the United States, the differences in tree communities found between different tidal várzea are attributed to slight differences in elevation (Cattanio et al., 2002), which influences flooding.

Climate change impacts

Tidal freshwater forested wetlands are generally adapted to tolerate short-term increases in salinity that arise from storm surges and/or decreased river flow during droughts, but their proximity to the coast also makes them prone to the chronic salinization driven by sea-level rise and land subsidence (Conner et al., 2007). The conversion of habitat from tidal freshwater to oligohaline manifests at sustained average annual porewater salinity around 2 g/l as trees die off and tidal swamps convert to brackish marshes or open water (Hackney et al., 2007). Tidal freshwater forested wetlands would be expected to respond to rising sea levels and the expanded "reach" of tidal influence by migrating upstream (Krauss et al., 2009). However, the extent of their migration is limited in many places by the presence of levees built for flood control, human development, and/or agricultural production (Doyle et al., 2010). By one modeling account, the extent of tidal freshwater forested wetlands in the southeastern United States is expected to decrease by 24–34 % by 2100 (Craft et al., 2009). It is likely that this habitat type is undergoing similar pressures globally with coastal development and sea-level rise.

Tidal saltwater forested wetlands (mangroves)

Mangroves are found in the intertidal zone (at the edge of continental land masses or islands) of low energy coasts. The hydrologic environment that mangroves occupy is similar to that of salt marshes, and the two habitat types are dynamic with regard to their shared boundaries. However, mangroves are more restricted globally due to their limited cold tolerance. Mangrove forests are most pervasive in tropical climates where they can readily outcompete salt marshes (Saintilan et al., 2009), but they are found in subtropical and warm temperate climates as well. In total, there are approximately 73 species and/or hybrids of mangroves (Duke et al., 1998) found in 123 countries, occupying roughly 13.7–15.2 M ha of intertidal, estuarine habitat worldwide (Spalding et al., 2010).

Mangroves occupy a wide variety of soil types, ranging from coralline soils that can be very alkaline (pH 8.5) and nutrient deficient to highly organic soils that can be very acidic (pH 5.8) and nutrient rich (Alongi, 2009). However, many mangrove forests are located on soils that are nutrient poor (Reef et al., 2010). Species distributions are often related to nutrient availability, and trade-offs exist between having morphological and physiological adaptations to tolerate low-nutrient environments and the ability to outcompete for dominance in high-nutrient environments (Krauss et al., 2008). Adaptations best suited for low nutrient conditions include the following: thick, carbon-rich evergreen leaves, efficient nutrient resorption prior to leaf fall, high biomass allocation to roots relative to shoots, and recolonization of previously used root channels (Reef et al., 2010); several *Rhizophora* species are well adapted for low-nutrient conditions. Mangroves that do well in high-nutrient conditions (e.g., *Avicennia* spp.) exhibit rapid growth, increased leaf area relative to stems and roots, thinner leaves with lower tannin concentrations, and greater photosynthesis and growth relative to the amount of water used (i.e., water use efficiency) (Krauss et al., 2008).

Hydrology and community composition

Mangrove forests have a remarkable zonation pattern relative to distance from open ocean water, with each zone often dominated by a single tree species and frequently sharp (but dynamic) boundaries between zones (Smith, 1992). This zonation pattern correlates with the frequency and duration of tidal immersion, which directly affects the degree of waterlogging (i.e., soil saturation), availability of nutrients, and salinity of the floodwater and soil porewater (Ball, 1988).

When a wetland is flooded, the oxygen available in the soil and water column is quickly depleted, resulting in oxygen deficiencies. Roots require oxygen for respiration using normal aerobic metabolic pathways, and many mangroves have structural adaptations to cope with oxygen deficiencies. Shallow root systems, extensive aerenchyma, and lenticels all increase the amount of oxygen available to roots (Ball, 1988).

Oxygen availability also affects soil nutrient availability. The availability of nitrogen and phosphorous, the two essential soil nutrients most widely linked to rates of plant growth, can change coincident with the amount of time the soil is flooded (see Mitsch and Gosselink, 2000). For instance, nitrogen that is bioavailable (able to be taken up by the roots and used for growth; i.e., nitrate or NO_3^-) reduces to biologically unavailable forms such as nitrite (NO_2^-), nitrous oxide (N_2O), dinitrogen (N_2), and/or ammonium (NH_4^+) with prolonged flooding. Phosphorus can become less bioavailable under anaerobic conditions by precipitating out with ferric iron, calcium, and aluminum, or binding onto clay particles, organic peat, and ferric and aluminum hydroxides and oxides. Because oxygen availability in the soil is dependent upon flooding frequency and duration, the amount of bioavailable nitrogen and phosphorous varies along the tidal immersion gradient. In general, mangrove growth is considered to be nitrogen limited in the anaerobic sediments at positions closest to the open ocean, while the availability of phosphorus limits mangrove growth in more oxidized soils at positions more landward (Boto and Wellington, 1983).

As facultative halophytes, mangroves are generally found in saline environments, but species exhibit a wide variety of growth responses to salinity. Optimal salinities for growth range from 2 to 26 g/l (5–75 % seawater), but most can also grow in freshwater (Krauss and Ball, 2013). All mangroves accumulate ions for osmoregulation but differ in the extent to which ions can be accumulated without metabolic dysfunction (Ball, 1988). There are four primary mechanisms by which mangroves cope with the overabundance of ions in saline water (see Krauss et al., 2008). The first is at the root, where mangroves exclude at least 90 % of external salts. The second is to store salt ions in vacuoles within the cells of the leaves. A third mechanism is to increase succulence of the leaf to maintain ionic balance (e.g., *Ceriops tagal*, *Xylocarpus* spp., *Osbornia* spp.). Finally, some species (e.g., *Acanthus*, *Aegialitis*, *Aegiceras*, *Avicennia*) excrete salt via glands on their leaves (Ball, 1988).

Climate change impacts

Mangroves are very susceptible to shifts in distribution that will result from increasing temperatures and sea-level rise. Rising air and ocean temperatures will likely increase mangrove growth rates and may allow them to expand their global distribution where they will encroach into salt marsh habitats (Traill et al., 2011). At the local scale, the effects of sea-level rise will vary depending on a variety of co-occurring environmental factors including tidal range, sedimentation and accretion (i.e., increase in soil surface elevation) rates, and local and regional subsidence. Biological feedback mechanisms, which vary depending upon species composition and growth rates, will also affect relative sea-level rise via sediment trapping and/or enhanced root growth (McKee, 2011). For instance, in many Australian mangrove forests, the rate of accretion equals or exceeds the rate of relative sea-level rise (Lovelock et al., 2011). Given continued seedling recruitment and adequate growth, the potential for landward migration of mangroves into new intertidal areas is strong. However, many mangrove communities occur near prime real estate areas; migration landward is restricted where human and natural barriers block their expansion.

Summary

Tidal freshwater forested wetlands are found along rivers at the uppermost extent of tidal influence, whereas mangroves are found nearest the ocean in the lowermost portion of the intertidal zone. Both systems are regulated largely by hydrology. Tidal freshwater forested wetlands will not persist where average annual porewater salinity exceeds 2 g/L, whereas mangroves can tolerate a wide range of salinities. However, mangroves are intolerant to freezing, and the adaptations necessary for mangroves to survive in saline conditions often limit their growth rates and ability to outcompete salt marsh species. Rising temperatures associated with climate change will likely expand the range of mangroves globally, at the detriment of salt marsh species in some cases. Sea-level rise is forcing the migration of both forested wetland habitats. Both forested wetland types have done this successfully, but levees and other barriers (natural and anthropogenic) will ultimately limit their expansion.

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Cross-references

Climate Change
Coastal Wetlands
Intertidal Zonation
Mangroves
Marine/Freshwater Mixing
Nutrient Limitation
Pneumatophores
Saltmarshes
Tidal Freshwater Habitat
Wetlands

FRINGING REEF

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Definition

A fringing reef is a shore-attached coral framework structure.

Description

Fringing reefs form close to shore, most often being shore attached, and are relatively thin veneers over non-reefal substrate (Steers and Stoddart, 1977). Fringing reefs can form as a continuous reef flat projecting from the shoreline or may be backed by a shallow lagoon sometimes termed a boat channel (Guilcher, 1988). For a reef to be classed as fringing the lagoon, it should have a maximum depth of 10 m (Milliman, 1974). Fringing reefs represent the first evolutionary stage of Charles Darwin's (1842) theory of coral atoll evolution when the coral reefs establish on the shoreline of a volcanic island in the open ocean. Gradual subsidence of the volcanic island leads to the fringing reef evolving into a barrier reef as vertical coral growth is maintained on the reef crest. Eventually, the central island sinks below the sea surface, leading to the formation of a coral atoll. The growth and form of a fringing reef is strongly influenced by the available accommodation space determined by the position of the antecedent surface on which the reef establishes in relation to sea level and the rate of sedimentation (Kennedy and Woodroffe, 2002). Fringing reefs may grow vertically to the sea surface in a keep-up or catch-up mode, having been established on a newly flooded substrate during a period of rising sea level. Fringing reefs may also initiate at the same elevation as a stable sea surface, prograding horizontally as a reef framework or over non-reefal sediments. Progradation can be characterized by progressive lateral accretion, but in some cases it may be episodic, occurring through the attachment of fore-reef coral bommies or patch reefs onto the reef front. In some cases fringing reefs may establish just offshore of a landmass either as a framework deposit or on top of an accumulation of storm-deposited rubble (Kennedy and Woodroffe, 2002).

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Cross-references

Artificial Reef
Oyster Reef

G

GEOHERITAGE

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Definitions

Geoheritage. The heritage value assigned to features of a geological nature encompasses globally, nationally, state-wide to regionally, and locally significant features of earth science that are intrinsically important or culturally important, offering information or insights into the evolution of the earth or into the history of earth science, or that can be used for research, teaching, or reference (Brocx, 2008). It encompasses the variety of rock types, stratigraphy, structural geology, geomorphology, and hydrology and covers a large variety of processes and products across a wide range of scales, from global tectonics, mountain building, and landscape evolution to local surface processes and products such as weathering, erosion and sedimentation, cliff faces, fossil sites and mineral localities, and, at the microscale, diagenesis and deformation.

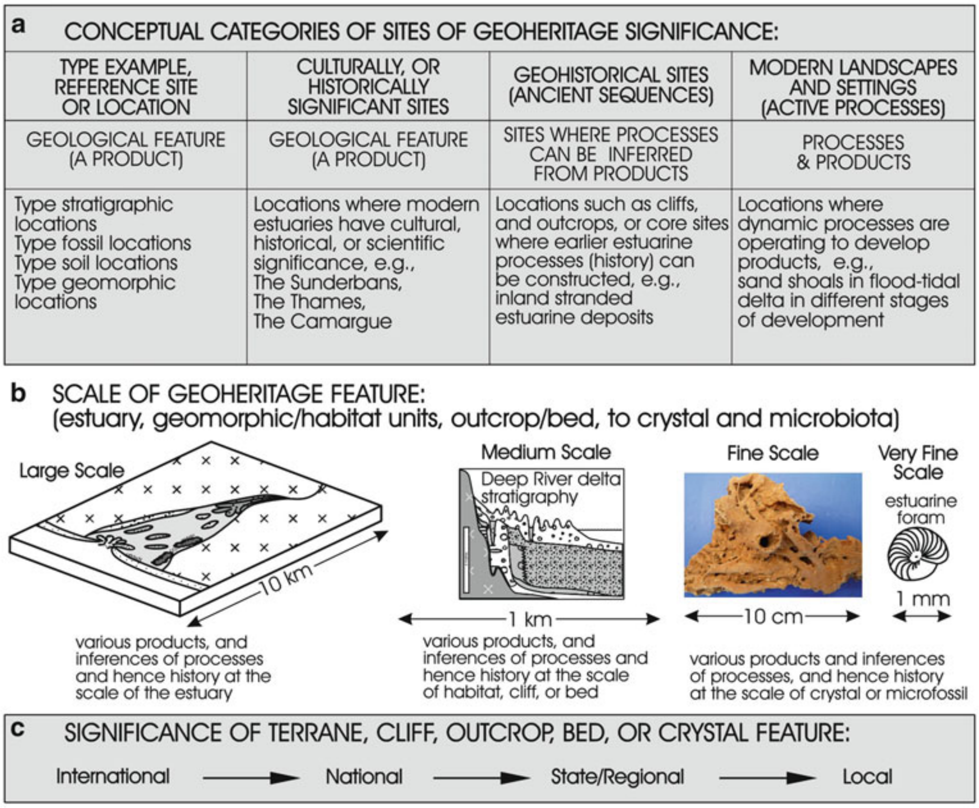
Geoheritage and geoconservation

Geoheritage and geoconservation are concerned with the preservation of earth science features and are important endeavors globally, as reflected in various international and intranational bodies set up for conservation, with agreements, conventions, and intergovernmental initiatives. Geoheritage includes igneous, metamorphic, sedimentary, stratigraphic, structural, geochemical, mineralogic, paleontologic, geomorphic, pedologic, and hydrologic attributes. This list of geological disciplines covers a large variety of processes and products, but, in addition, it also traverses a wide range of scales, from global tectonics, mountain building, and landscape evolution to local surface processes and products such as weathering, erosion and sedimentation, cliff faces, fossil sites and mineral localities, and, at microscale, diagenesis, crystal defects, and deformation, among others. The scope of geoheritage is summarized in Figure 1. It involves assigning geological features or sites of geoheritage significance to one of four categories, assigning a scale of reference to the sites and evaluating their level of significance.

While geoheritage relates to the heritage of features of a geological nature, *geoconservation* is the action that works toward the preservation of sites of geoheritage significance for purposes of heritage, science, or education, i.e., preserving sites of geoheritage significance once the level of their significance has been determined. Geoconservation thus can encompass all important geological features from the regional scale to the individual crystal. Geoconservation can involve preservation of specific sites (special sites) or of geological ensembles. The former are where a significant geological feature occurs in isolation or may have historical or cultural significance. Sites are formally identified in the British Isles as (geological) Site(s) of Special Scientific Interest (SSSI) or Regionally Important Geological/Geomorphologic Sites (RIGS) (Ellis et al., 1996).

In regard to estuaries, there has been emphasis on their biotic or biological significance, for instance, in terms of their vegetation complexes, their productivity, and their fisheries and hence their conservation and management, and less on the importance of their abiotic significance, i.e., geological, sedimentological, hydrological, and evolutionary attributes. Geoheritage and geoconservation can also be directed to the recognition and preservation of the abiotic realms of estuaries. For instance, it can involve the recognition and geoconservation of

M.J. Kennish (ed.), *Encyclopedia of Estuaries*, DOI 10.1007/978-94-017-8801-4,



a CONCEPTUAL CATEGORIES OF SITES OF GEOHERITAGE SIGNIFICANCE:

TYPE EXAMPLE, REFERENCE SITE OR LOCATION	CULTURALLY, OR HISTORICALLY SIGNIFICANT SITES	GEOHISTORICAL SITES (ANCIENT SEQUENCES)	MODERN LANDSCAPES AND SETTINGS (ACTIVE PROCESSES)
GEOLOGICAL FEATURE (A PRODUCT)	GEOLOGICAL FEATURE (A PRODUCT)	SITES WHERE PROCESSES CAN BE INFERRED FROM PRODUCTS	PROCESSES & PRODUCTS
Type stratigraphic locations Type fossil locations Type soil locations Type geomorphic locations	Locations where modern estuaries have cultural, historical, or scientific significance, e.g., The Sunderbans, The Thames, The Camargue	Locations such as cliffs, and outcrops, or core sites where earlier estuarine processes (history) can be constructed, e.g., inland stranded estuarine deposits	Locations where dynamic processes are operating to develop products, e.g., sand shoals in flood-tidal delta in different stages of development

c SIGNIFICANCE OF TERRANE, CLIFF, OUTCROP, BED, OR CRYSTAL FEATURE:

International → National → State/Regional → Local

Geoheritage, Figure 1 The scope of geoheritage of estuaries.

end-member types of estuaries as global "type examples" of the variety forms expressed around the world in response to climate, hydrodynamic setting, sediment types, and framework geology. At this scale, geoheritage recognizes the range of estuarine systems that are manifested around the globe and attempts to address the significance of the variety of these estuaries that have formed in different geological, hydrological, sedimentological, and climatic settings within a variable biogeographic context. At the next level, geoheritage and geoconservation can involve the geoconservation of geological processes and products operating and occurring within estuaries, e.g., deltaic sedimentation and its variety of landforms, sand platforms and their surface bedforms, evolution of estuarine stratigraphy, stratigraphic/hydrologic interactions, and styles of hydrochemical mixing. At the finest scale, geoheritage and geoconservation can involve the recognition and geoconservation of microscale processes and products, often specific to an environmental setting and climate, e.g., diagenetic features such as calcitization of shoreline rush rhizomes, occurrence of dolomite, formation of pyrite nodules, the permineralization of skeletons, and the effects of freshwater seepage.

Brocx and Semeniuk (2007) assign sites of geoheritage significance to one of four categories (Figure 1): (1) reference sites and/or type locations, (2) sites of cultural or historical importance, (3) geohistorical sites showing ancient sequences where earth history can be determined, and (4) modern landscapes and settings where earth processes are still active. Some estuaries, of course, may belong to more than one category.

In regard to *reference sites and/or type locations* (Category 1) for estuaries, once they have been classified as to a type, the reference locations of end-member estuary type or the best example of an estuarine type can be identified and allocated as an international or national heritage locality. In this context, for comparisons of estuaries for geoheritage evaluation, it is important to have a worldwide applicable estuarine classification and nomenclature that can be used systematically and comparatively to differentiate estuary types based on landform/coastal setting, shape and size of estuary, tidal and wave regime, sediment assemblages, seawater/freshwater mixing style, climate, and biota. A selection of estuaries that stand out globally as distinct and geomorphically significant because of either their size, internal landforms, representativeness, or naturalness and that can be used as estuarine reference sites and/or type locations are Lake St Lucia (Natal, South Africa), Solway Firth (Scotland), Gironde estuary (France), the Elbe (Germany), the deltaic complex of the Ganges-Brahmaputra (Sundarbans National Park, India), Walpole-Nornalup Inlet Estuary

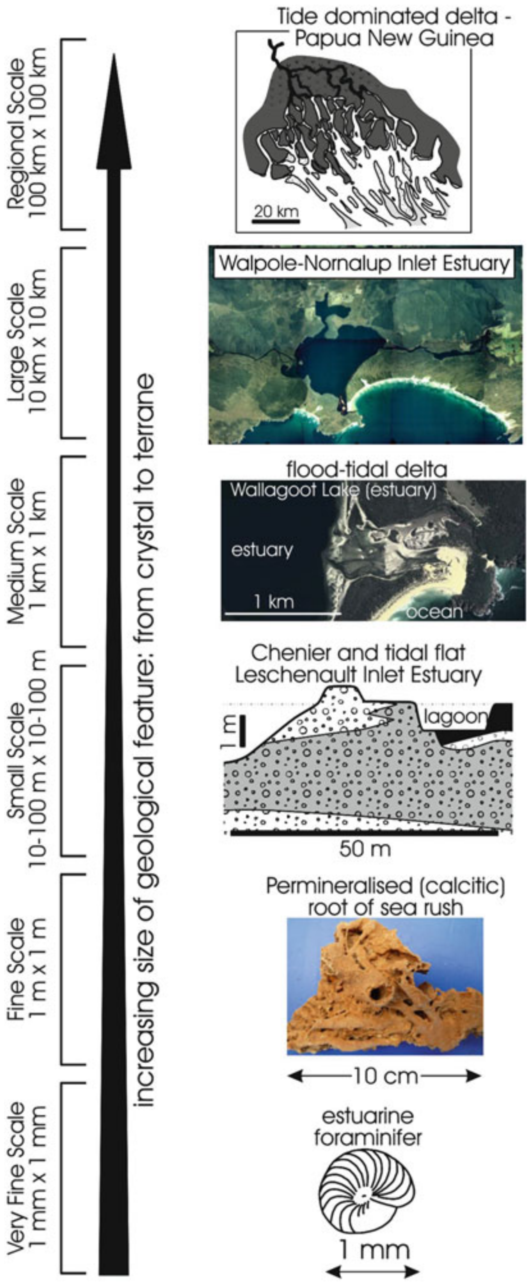
(Western Australia), Fitzroy River estuary (Queensland), Gulf of Saint Lawrence (Canada), Chesapeake Bay (North America), and the Amazon estuary (Brazil).

In regard to the *cultural or historical significance* (Category 2), estuaries may function as highly significant systems to the local traditional owners or may carry historical significance. The Camargue in the estuary of the Rhone (France) is an example of the former, and Port Hacking (Australia) and the Thames (the United Kingdom) are examples of the latter.

Estuaries can also function as *geohistorical sites* (Category 3) showing ancient sequences where earth history can be determined. In this regard, they hold records over the past 7,000 years when sea level stabilized to its present position of coastal history and valley-fill history in their stratigraphy and stranded estuarine geomorphology (Roy, 1984). The estuarine sequences recording estuarine evolution documented in Australia and North America (Fisher, 1969; Roy, 1984) are examples of the geohistorical importance of estuaries and illustrate the variety of pathways that an estuary may take in its development. Estuaries also hold records over the Pleistocene of previous estuarine history in their stratigraphy and older estuarine geomorphology.

Estuaries are excellent examples to illustrate *modern landscapes and settings* (Category 4) where earth processes are still active. It is an environment where fluvial sedimentation interacts with the estuarine basin; intra-estuarine processes mobilize and deposit sediments into shoals, platforms, and basin-fill sheets; flood and ebb tides develop flood and ebb tidal deltas; and a plethora of biological, geochemical, hydrochemical, and physical processes at the finest scale result in a plethora of sedimentary bedforms, sedimentary structures, sedimentary deposits, biogenic deposits, and mineral precipitates.

Scale is important to consider in geoheritage/geoconservation since features of significance can range from landscapes and phenomena at montane scale to outcrops and bedding planes; to that of crystals, i.e., regional, large, and medium; and to small, fine, and very fine scales. In many locations, estuary sites are important because of crystal-sized phenomena and crystal fabrics (e.g., dolomite or permineralization in estuarine shorelines) and at the next scale, because of outcrops and bedding scale features, such as elevated estuarine fossil sites (Brocx, 2008). Important geological/geomorphological features continue to occur in an increasing scale, right up to the scale of mountain ranges and major drainage basins which, in the case of estuaries, involves the size of the estuarine embayment which may be tens of kilometers in size and can involve the lower reaches of the drainage basin. On the other hand, there may be small-scale estuaries that are merely tens of meters in width. Scales of features of geoheritage significance in estuaries are illustrated in Figure 2.



Geoheritage, Figure 2 Scales of geological features in estuaries.

The level of importance attributed to a given feature of geoheritage significance is related to how frequent or common is the feature within a scale of reference and/or how important is the feature to a given culture. Five levels of significance are recognized (Brocx and Semeniuk, 2007; Brocx, 2008): (1) international (one of, or a few, or the best of a given feature globally), (2) national (though globally relatively common, one of, or a few, or the best of a given feature nationally or continentally), (3) state-wide to regional (though globally relatively common and occurring throughout a nation or a continent, one of, or a few, or the best of a given feature nationally/continentally or regionally), and

(4) local (occurring commonly through the world, as well as nationally/continentally to regionally, but especially important to local communities). Levels of significance of geoheritage features of (and in) estuaries are illustrated in Figure 3.

Geodiversity and geoparks

It is important to note that just as biological systems are diverse, geological systems are also diverse (geodiversity), as are estuarine systems. For instance, using analogues of "wetlands" and "dunes" to illustrate geodiversity, the terms "wetland" and "dune" carry implications of a large array of wetland types and dune forms. Thus, wetlands may be lakes, sumplands, damplands, paluslopes, and palusmonts (Semeniuk and Semeniuk, 1995) or in the terms of the Ramsar Convention Bureau (1991), fens, marshes, ponds, oases, etc. Dunes may be linear, parabolic, hairpin, star, or barchan, among others. Estuaries carry the same degree of geodiversity, as exemplified by variation in their setting, shape, size, estuarine landforms, hydrology, sedimentary processes, diagenesis, and internal functioning and can be classified as estuarine types. They also carry geodiversity in their history as expressed in their geomorphology, paleogeomorphology, and stratigraphy. Similar to the objective of nature conservation, to conserve the vast diversity of life forms, an objective of the conservation of sites of geoheritage significance in estuaries would be the conservation of the variety of their forms on the earth. In this context, the conservation of a single "estuary" as an example of an estuarine system as representative of the full variety of estuarine types globally is insufficient (or to use the "dune" analogue, the conservation of a single parabolic dune form in one area as representative of the full variety of dune types globally is insufficient). If estuaries, for instance, exhibit a large diversity of geometric and hydrologic types, stratigraphic fills, and origins, then at the least, their conservation should encompass an example of each of the types extant.

Large sites of geoheritage significance or an amalgamation of numerous smaller sites of geoheritage significance can be assigned to geopark status where there is geoconservation of an ensemble of geological features. The Global Geoparks initiative supported by UNESCO defines a geopark as a territory encompassing one or more sites of scientific importance, not only for geological reasons but also by virtue of its archaeological, ecological, or cultural value. As such, this type of geopark crosses discipline boundaries and integrates geological, archaeological, ecological, or cultural endeavors. The European Geoparks Network, established in 2000 (Zouros, 2000), defines a geopark as an area to conserve and valorize geological heritage through the integrated and sustainable development of their territories. The Asia Pacific Geoparks Network, founded in 2007, defined a geopark as a nationally protected area containing a number of geological heritage sites of particular importance, rarity, or aesthetic appeal. These earth heritage sites are part of an integrated concept of protection, education, and sustainable development. All these initiatives aim to protect geodiversity, promote geological heritage, and support local sustainable economic development, thus involving community and commercial interests.

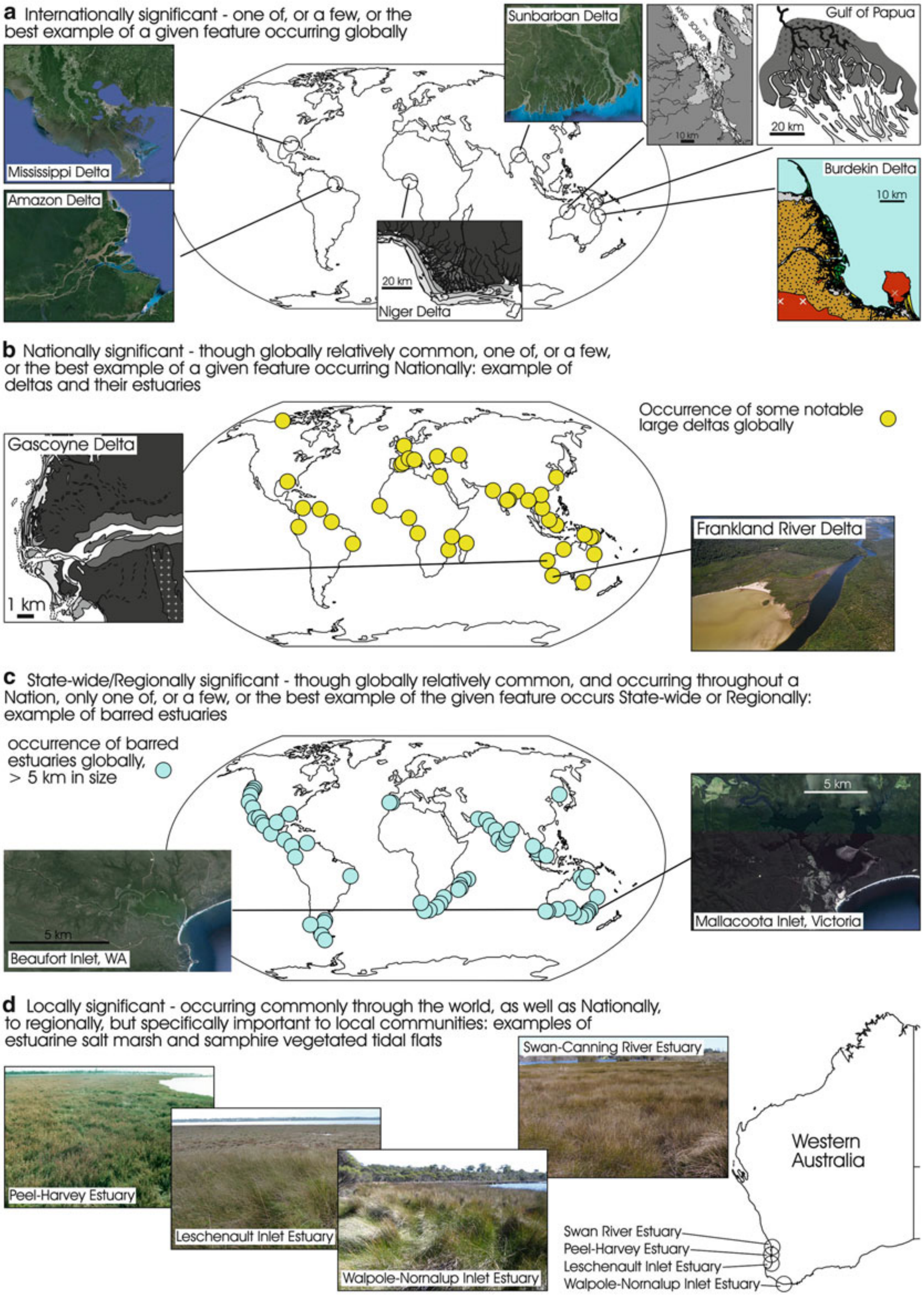
Estuaries lend themselves to allocation as geoparks because they inherently have multiple use (fishing, boating, shoreline nature walks, areas of conservation for waterbirds) and often illustrate landscape and geology that can be used for science and education. Also, as the interface between marine and freshwater, with a multitude of processes operating internally, in terms of geomorphology, sedimentology, and microscale features, they provide a wealth.

Brocx and Semeniuk (2009) valorized the Walpole-Nornalup Inlet Estuary in Western Australia and identified it as a potential geopark, wherein the various Cainozoic and Holocene geological features could be used as features for nature tours. Thus, estuaries can be viewed as geoparks, i.e., as conservation, promotional entities focused on geological and geomorphological attributes for local sustainable development. To provide a comparative example, using a biological analogue, if a region can be conserved for its biological attributes and biodiversity, and called a "Nature Reserve," the same rationale can be applied to areas manifesting ensembles of interrelated significant geological and geomorphological features. The former can be considered to be worthy of conservation as a "biopark," and the latter can be considered to be worthy of conservation as a "geopark." Once protected in conservation parks, both can be utilized for local socioeconomical sustainable development through ecotourism or geotourism (i.e., as "biotours" or as "geotours") and for science and education. With estuaries, the ensemble and interacting biological and geological features underscore the principles that biological systems are, in part, determined by abiotic factors and that "biodiversity is underpinned by geodiversity."

Summary

Geoheritage and geoconservation are important management and conservation endeavors that can and should be applied to estuaries and their internal functioning and environments. While there has been emphasis to date on the biotic or biological significance of estuaries, e.g., in terms of their vegetation complexes; their productivity, fisheries, and avifauna; and hence their conservation and management, there has been less focus on the importance of their abiotic significance, i.e., their geological, sedimentological, hydrological, and geo-evolutionary attributes for purposes of geoheritage and geoconservation and for their allocation and inscriptions as geoparks.

Geoheritage and geoconservation can be directed to the recognition and preservation of the abiotic realms of estuaries. For instance, estuaries of geoheritage significance can be assessed as to their global, national, regional, or



Geoheritage, Figure 3 Levels of geoheritage significance of geological features of and within estuaries and deltas (based on the approach described by Brocx and Semeniuk (2007) but modified to focus on estuaries and deltas). The examples used to illustrate nationally significant geoheritage features are deltas (and the estuaries at their mouths) shown in a globally comparative manner. The examples used to illustrate state-wide/regional significance are barred estuaries (>5 km in size), drawn from Australia and are shown in a comparative global and national context. The examples used to illustrate local significance are tidal saltmarsh platforms shown in a context of barred estuaries of southeastern Australia shown in a context of barred estuaries in Australia.

local level of significance on geological values and recognized as such. Estuaries that stand out globally as distinct or geomorphically significant and that can be used as estuarine reference sites include Lake St Lucia, the Gironde estuary, the deltaic complex of the Ganges-Brahmaputra, Walpole-Nornalup Inlet estuary (Western Australia), Fitzroy River estuary (Queensland), Gulf of Saint Lawrence, Chesapeake Bay, and the Amazon estuary. Aside from those of Global significance, many other estuaries, geologically, qualify to be allocated to national significance.

Estuaries readily conform to being allocated to one of the categories of geoheritage significance, viz., as reference sites and/or type locations, as those of cultural or historical importance, as geohistorical sites, or as sites where earth processes are still active, as many stand as classic examples of the type of estuary they represent and are quite active today as interactive riverine-to-marine systems and also steeped in human history. Since geoheritage encompasses a wide range of disciplines within the earth sciences and involves geological features at all scales, from the scale of large estuaries to the microscopic, there is potential to involve many natural phenomena within estuaries. At the largest scale, the range of end-member estuary types and their climate setting may render an estuary to be a geoheritage feature of global significance. At the smallest scale, the precipitation of exotic minerals or the mineralization of plants in the tidal zone may render an estuarine environment also as a geoheritage feature of global significance.

Estuaries carry a large degree of geodiversity, exemplified by variation in their setting, shape, size, estuarine landforms, hydrology, sedimentary processes, diagenesis, and internal functioning and can be classified as to many estuarine types. They also carry geodiversity in their history as expressed in their geomorphology, quaternary paleogeomorphology, and stratigraphy. As such, they lend themselves for use as geoparks where Quaternary history and Holocene geological processes and landscape evolution can be demonstrated.

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Cross-references

Determining Geoheritage Values

GEOMORPHOLOGICAL MAPPING

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Definition

Geomorphological mapping refers to the process of defining, identifying, and graphically delimiting the fundamental geomorphic units that comprise a landscape.

Principles and background

Geomorphological maps typically are developed for geographically small areas between 10^1 and 10^4 km^2 at scales between 1:5,000 and 1:50,000. Despite this relatively narrow range of areal and scale parameters, a hierarchy of different geomorphic features can be identified as basic units (Knight et al., 2011; Smith, 2011). Map units typically reflect the intrinsic characteristics of the landforms (i.e., morphology, morphometry, chronology, and formation processes) and related non-geomorphological data regarding the geographic, geologic, or environmental setting (Pavlopoulos et al., 2009). Many forms and processes, such mass wasting, can be identified over a wide range of scales. However, at small scales, small features and processes are difficult to identify and delimit, while large features, which cannot be delimited at large scales, are more obvious. Therefore, the intended purpose of the map dictates the smallest forms or processes that should be depicted.

Historically, the cartographic manifestations of geomorphological mapping efforts were largely symbolic maps. By the mid-twentieth century, improvements in the accessibility of aerial imagery and topographic data led to an increased interest in, and usefulness of, geomorphological maps (Verstappen, 2011). However, unique sets of map symbols, developed independently by different researchers, made it difficult to compare geomorphological maps, even when developed for similar terrains.

International efforts in the mid-twentieth century to standardize geomorphic mapping techniques and symbols resulted in some improved consistency in the presentation of content (see the IGU Unified Key (1968) and St. Onge (1968) for standardized symbol sets from that period). Despite this effort, departures from the standard representation of information on geomorphic maps continue because of the proliferation of digital data used to produce specialized maps for diverse purposes (i.e., hydrology, land-use planning, disaster and hazard management, and land-surface evolution).

In recent years, the increased availability of high-resolution remotely sensed data has shifted the balance of mapping efforts from symbolic depictions on traditional paper maps to the realm of modeling and geographic information system (GIS) software (see Proceedings of 41st Binghamton Geomorphology Symposium, James et al., 2012). Modern geomorphic mapping often relies on the mathematical and statistical analyses of thematic data covering the same geographic domain at multiple spatial and temporal resolutions. Like a traditional paper-based geomorphological map, digital geomorphic mapping allows for the organization of geomorphic data into hierarchical classes with spatially and temporally variable properties and geometric relationships. However, digital geomorphological maps can provide a more complete geometric and multi-scale description of landforms and processes via generalization and decomposition procedures that emphasize unique aspects of the landscape.

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GLACIATED ESTUARINE SYSTEMS

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Synonyms

Fjord

Definition

Glaciated estuarine systems are inundated transition zones between rivers and marine waters in areas that have been directly modified by glaciers.

Physiography

Glaciated estuarine systems may refer to both (1) narrow, overdeepened, steep-sided glacial troughs generally referred to as fjords and (2) eustatically drowned low-relief landscapes once covered by glaciers. While fjords display pronounced relief characteristics from glacial erosion, the effects of a glacier on the estuary may also be depositional. For example, many estuaries in the Pacific Northwest, northeastern North America, and northern Europe are characterized by low-relief landforms and/or sediments directly deposited by Pleistocene glaciers.

Cross-references

Fjord

HABITAT COMPLEXITY

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Synonyms

Architectural complexity; Habitat architecture; Habitat heterogeneity; Substrate heterogeneity; Topographical complexity

Definitions

Krebs (1994) has defined a habitat as any part of the biosphere where a particular species can live, either temporarily or permanently.

Complexity is defined as one aspect of habitat quality that is thought to influence the size, structure, distribution, and stability of populations.

Habitat complexity refers to the level or strength of interaction between a species and its environment.

Introduction

In estuarine ecosystems, habitat complexity is one of the most important factors structuring biotic assemblages. It is frequently an important variable affecting species abundance and diversity. Habitats with high habitat complexity typically support more species and individuals than nearby less complex habitats. Identifying the key aspects and spatial scales of habitat complexity is critical to understanding the ecology and conservation of a range of communities (Gratwick and Speight, 2005; Kovalenko et al., 2012).

Description habitat complexity

Habitat complexity has been described in terms of structural components. Most authors only examine one or two variables out of at least six that are recognized (Gratwicke and Speight, 2005). These six include the following: (1) topographic complexity or rugosity of the substratum; (2) substratum diversity; (3) variety of refuge hole sizes; (4) vertical relief or height of substratum architecture; (5) percentage live cover, including corals and seagrasses; and (6) percentage hard substratum.

Topographic complexity or rugosity of the substratum

This component can be measured in several ways. The most frequently used measurement of reef surface complexity is rugosity. This is most commonly measured by the chain-and-tape method whereby a ratio of the length of a chain draped across the reef surface to the linear stretched length is calculated. This ratio, hereafter referred to as linear rugosity, provides a rugosity measurement (index) that has been used to determine its relationship with fish abundance, biomass, species richness, and species diversity indices (Walker et al., 2009).

Substratum diversity

The substratum diversity is the percentage cover of different substrata on a 10-m transect and is measured from stereophotographs. The number of centimeters of the tape measure covering each substratum type, expressed as a percentage of the total transect length, gives an unbiased estimate of their cover. Hard and soft corals are classified into recognizable taxonomic units for this analysis. A second measure of the complexity of the substratum along the transects, the Shannon-Wiener index (H’), is calculated from the percentage cover data of different substratum types from each depth to give a measure of biological diversity (Roberts and Ormond, 1987).

M.J. Kennish (ed.), *Encyclopedia of Estuaries*, DOI 10.1007/978-94-017-8801-4,

Variety of refuge hole sizes

Holes are very important to small fish on coral reefs as refuges from predators. The number of holes is also counted within the same five 1 m^2 quadrats on each transect from which surface indices are calculated. They are classified into the following size ranges (entrance diameter): 1–5, 6–15, 16–30, 31–50, and >50 cm (Roberts and Ormond, 1987).

Vertical relief or height of substratum architecture

A surface can be topographically complex in a number of ways, and there are correspondingly many ways of measuring such complexity. For comparative studies, variation in complexity is usually described with a univariate index. Researchers have a choice of indices with complexity values typically calculated from a transect across the surface. There are essentially two ways in which the complexity of the surface can be translated into an index. The distance traveled along the surface compared to the linear distance between the ends of the transect gives a measure of the extra surface introduced by following cracks and protrusions on the surface. Typically chains are used to follow the surface profile and calculate this type of index. As an alternative to surface following techniques, some form of measurement of the surface can be taken, with statistics calculated from the resulting profile. Typically these surface measurements involve recording the different heights of pins in a profile gauge or the use of a stereophoto to reconstruct heights. Statistics calculated from these profiles include the sum of consecutive height differences between horizontally adjacent points and indices based on the variance of change in slope angle between consecutive measurements. Separate indices measure slightly different properties of a surface but tend to be correlated for the same profile (Frost et al., 2005).

Percentage live cover, including corals and seagrasses

Percentage live cover throughout the area is visually assessed along 50 m of reef around the perimeter of a basin at points on a 1-km grid. Cover is estimated using the following scale: 0 % (dead), <2 %, 2 to <5 %, 5–10 %, and >10 %. This method of measuring coral cover is preferred to more quantitative techniques due to the limited time available and because data from abundance categories have proved reliable in quantifying changes in other assemblages (Bell and Galzin, 1984).

Percentage hard substratum

Hard substratum refers to the percentage of substratum that is not mud, sand, or rubble (Gratwick and Speight, 2005).

How to measure the complexity of the habitat

Understanding the relationships between species biological traits and the environment is crucial to predicting the effect of habitat perturbations on communities. Habitat complexity is a multivariate problem; one of the characteristics of environmental data is the complex relationships that exist between them. In order to elucidate the relationships between biological assemblages and the environmental variables, multivariate tests can be applied: canonical correspondence analysis (CCA), multiple regression analysis, non-metric multidimensional scaling (nMDS), multivariate analysis of variance (MANOVA), and others.

Many studies have assessed local effects of habitat complexity, reporting increased richness and abundance in more complex habitats. Greater habitat complexity is often associated with a greater abundance and diversity of organisms. High complexity habitats may reduce predation and competition, thereby allowing more individuals to occupy a given area. However, the relationship between habitat complexity and species diversity is still unclear. In essence, quantitative indices of habitat complexity are available for most habitat types and should be more widely employed in future studies to better understand the mechanistic role of habitat structure. Some of these indices can be developed further to enable consistent comparison among different habitats, thus alleviating the problem of the lack of integration between fields (Kovalenko et al., 2012).

Summary

The analysis of habitat complexity is an important aspect of understanding the dynamics of estuarine communities. This analysis should consider a multivariate approach, as the habitat is determined by the synergy of environmental factors such as topographic complexity or rugosity of the substratum, substratum diversity, variety of refuge hole sizes, vertical relief or height of substratum architecture, percentage live cover (including corals and seagrasses), and percentage hard substratum. Furthermore, it is necessary to develop and implement the use of new indices that help discern more clearly the relationship of diversity to habitat complexity.

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Cross-references

Environmental Gradients
Estuarine Habitat Restoration
Habitat Loss
Shannon-Weaver Diversity Index

HABITAT LOSS

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Synonyms

Habitat clearing; Habitat conversion; Habitat destruction; Habitat reduction

Definition

Airoldi and Beck (2007) have defined habitat loss as a measurable reduction in the abundance and distribution of natural habitats. Habitat loss is a term that is now in widespread use because of the ongoing impacts on natural systems associated with human population growth and the increasing need for development space and natural resources (Millennium Ecosystem Assessment, 2005). At the same time, the term "habitat loss" is often defined in inconsistent ways. Most often "habitat" refers to the predominant features that create structural complexity in the environment, such as plants (e.g., pine forests, sea grass meadows) or animals (e.g., coral reefs), and "loss" indicates a focus on a measurable reduction in habitat cover (i.e., areal extent). Loss clearly occurs when natural habitats, such as forests, are cut down and slated for construction of buildings and roads or when salt marshes are filled with sediments and blocked from the sea to form agricultural fields. Sometimes the loss of one habitat can mean the gain in another habitat such as when seagrasses die from disease outbreaks and sand or mudflats remain. Even in these cases of transition, the common transition is from a more to a less structurally complex habitat. It is very rare for an area to transition from a less to a more structurally complex habitat unless there is active habitat restoration. In the oceans, this widespread phenomenon has been referred to as the flattening of the marine bottom.

Habitat degradation (alteration or change) often is a precursor to the loss of natural habitats. However, degradation is difficult to measure because it represents a change in condition, not a change in distribution (i.e., habitat loss). Habitat fragmentation falls between loss and degradation (Bender et al., 1998). Fragmentation occurs when previously continuous habitats become patchy or less dense (e.g., loss of patches of trees within a larger forest). The distinction between loss and fragmentation is generally one associated with monitoring resolution because many large-scale surveys and spatial imagery do not capture increases or decreases in patchiness.

The term habitat loss is most commonly used to refer to the destruction of natural habitats as a direct or indirect consequence of many human activities, including urban development, organic and inorganic pollution, and overexploitation of resources. Habitat loss, however, can also occur as a result of natural events such as floods, droughts, volcanic eruptions, earthquakes, and climate fluctuations.

Habitat loss is ranked among the leading causes of species extinction both in terrestrial and marine systems (Pimm and Raven, 2000; Airoldi et al., 2008). Indeed, habitat loss results in the immediate extirpation of local populations of resident species and the loss of the ecosystem functions and services that these populations provided (Millennium Ecosystem Assessment, 2005).

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Cross-references

Estuarine Habitat Restoration
Habitat Complexity

HALOCLINE

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Definition

A halocline is a vertical zone in the water column in which salinity changes rapidly with depth (Halocline, 2013).

Description

Haloclines in the ocean are either seasonal or permanent. In coastal inlets, lagoons, fjords, and poorly mixed estuaries, haloclines form where freshwater overlies seawater. A vertical salinity gradient in haloclines can reach 46 PSU m^{-1} (50-cm layer of Fjord Hunnbunn, Norway; Yakushev et al., 2013).

Because salinity (together with temperature) affects the density of seawater, it can play an important role in its vertical stratification. Pycnoclines (layers of rapid change of density with depth) are often coupled to haloclines and thermoclines (layers where the temperature gradient is greater than that of the warmer layer above and the colder layer below).

The formation of a pycnocline restricts vertical mixing and leads to oxygen depletion in deeper waters and the formation of anoxia. This process occurs, for example, in the Black Sea, many marine deeps (e.g., Gotland Deep in the Baltic Sea), anoxic fjords, and some deeper estuaries.

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Cross-references

Anoxia, Hypoxia, and Dead Zones
Oxygen Depletion

HALOGENATED HYDROCARBONS

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Synonyms

Halocarbons

Definition

Halogenated hydrocarbons are chemical compounds containing carbon and hydrogen atoms and one or more halogens (bromine, chlorine, fluorine, iodine) in their chemical structure to form organobromine, organochlorine, organofluorine, and organoiodine compounds. The organochlorine compounds (i.e., chlorinated hydrocarbons) are a particularly well-known and hazardous group of halogenated hydrocarbons in estuarine and marine environments. As a group, the halogenated hydrocarbons consist of some of the most persistent, ubiquitous, and toxic environmental pollutants. They commonly originate from agricultural activities, industrial operations, and urbanization (Kennish, 1997; Kennish, 2001; Kennish et al., 2008).

Description

Among the wide array of halogenated hydrocarbons found in estuarine and marine environments, the higher-molecular-weight compounds pose the most serious threat to estuarine and marine organisms. This is so because the higher-molecular-weight halogens tend to accumulate in biota (notably in lipid-rich tissues) and can then concentrate up food chains. The organochlorine contaminants (e.g., DDT, chlordane, toxaphene, PCBs) are noteworthy because they have unique properties that can increase their negative impacts. For example, they are chemically stable, highly mobile and generally toxic, hydrophobic, resistant to degradation, and persistent in the environment. In addition, they have an affinity for living organisms, are lipophilic, and exhibit considerable bioaccumulative capacity. They may also biomagnificate through food webs. Hence, organisms at the uppermost trophic levels (i.e., dolphins, porpoises, seals, whales, and other mammals) typically carry the highest contaminant residues (Kennish, 1997; Kennish et al., 2008).

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Cross-references

Anthropogenic Impacts
Nonpoint Source Pollution
Oil Pollution
Polycyclic Aromatic Hydrocarbons
Trace Metals in Estuaries

HEADLAND BREAKWATERS

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Synonyms

Bay beach; Embayment; Pocket beach

Definition

Headland breakwaters are coastal structures placed roughly parallel to the coast and attached to the land with a sand spit (i.e., tombolo), thereby forming a headland.

Introduction

Headland breakwaters are structures, often made of rock, that can occur as a single breakwater or a series of breakwater that create a series of pocket beaches (Figure 1). These types of structures are used worldwide for the protection of the coast and to stabilize beach nourishment. Much of the research and applications have centered on open ocean applications and the Great Lakes (Suh and Dalrymple, 1987). The use of headland breakwaters in estuarine systems has been limited except for the United States' mid-Atlantic coast. The Chesapeake Bay, in particular, has numerous examples of this application (Hardaway and Gunn, 2010; Hardaway and Gunn, 2011). This is due in part, to a long history of installations where beach habitat can be considered as a trade-off for encroachment onto state-owned bottom.

The Chesapeake Bay estuarine system like many others in the United States and around the world consists of a variety of shorelines that vary from low, upland banks and marshes to beaches and dunes to high bluffs. Erosion of these shorelines becomes significant when fetch exposure, the distance over open water that wind can blow and generate surface waves, exceeds a few kilometers. Erosion becomes severe when shorelines are exposed to fetches exceeding 16 km or more. Critical erosion has been defined as erosion that immediately threatens upland improvements and infrastructure no matter what the fetch exposure. The use of headland breakwaters coupled with beach fill to create stable pocket or embayed beaches for shoreline management has become somewhat commonplace. Over the last 25 years, numerous research and project installations have paved the way for widespread usage (Hardaway and Gunn, 1991; Hardaway et al., 1995; Hardaway and Byrne, 1999; Douglass and Pickel, 1999; Hardaway and Gunn, 1999a; Hardaway and Gunn, 1999b; Hardaway and Gunn, 2002).

Properly designed and installed headland breakwaters with beach fill for pocket beaches and wetlands plantings provide shore protection and create a "full" coastal profile with three components, the beach, backshore, and dune, which enhances habitat, creates a tertiary buffer for upland runoff and groundwater, and provides access for recreation. Establishing vegetation zones within the headland breakwater system is a critical design element since wetlands grasses also create an erosion-resistant turf during storm events. Along the existing beaches on the open bay and broad rivers of the Chesapeake Bay estuarine system, dune grasses can only survive above a stable berm. Intertidal grasses must reside in sheltered regions. This coastal profile also accommodates environmental permitting requirements of habitat enhancement.

Design considerations

Perhaps the most important parameters in headland breakwater design are the width and elevation of the beach in the gaps of the breakwater system (i.e., minimum bay beach size). The beach must be high and wide enough to offer protection, usually for the base of a graded upland bank, under design storm conditions. Design storms are at least the 25-year event, and the breakwater itself should withstand the 100-year or greater storm.

In headland breakwater design, the beach morphology emulates nature so the beach profile should be assessed first when designing any headland breakwater system. The design of the beach begins with establishing the minimum design beach width (Bm) and profile in the context of a stable pocket beach that will be held between headland breakwaters (Figure 2). This will determine the amount of beach nourishment required. With Bm established, the breakwater length (Lb), the breakwater gap (Gb), and the bay indentation distance (Mb) are determined and depend on the wave environment. Hardaway et al. (1991) developed relationships between these design parameters. In particular, the statistical relationship between breakwater gap (Gb) and the bay indentation distance (Mb) should be 1.65:1. Hardaway and Gunn (2002, 2010) further assessed this relationship and found it to be a usable first step in the design process.

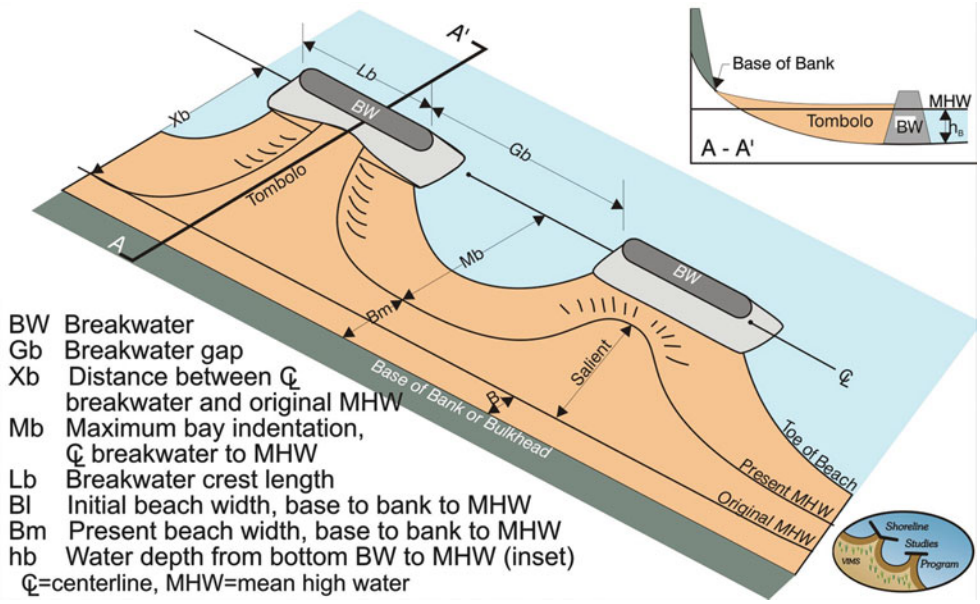
Numerous studies, as documented in Chasten et al. (1993), show that as a breakwater is lengthened relative to its distance offshore, a tombolo is more likely to develop. A tombolo is an essential element in headland breakwater systems. The tombolo must be developed with the addition of beach nourishment in Chesapeake Bay projects since natural sand supply generally is limited. It appears that as breakwater length approaches the design wave length by twofold, it can better hold a tombolo, particularly when the breakwater acts as a headland in a multiple breakwater unit system. The level of tombolo attachment may vary from attachment above high water to a low water connection.

Suh and Dalrymple (1987) demonstrated that when the gap between two diffraction points (i.e., the ends of adjacent breakwaters) becomes approximately twice the incident wave length or more, the shoreline behind each breakwater responds independently as if there is no interaction among breakwaters. This mechanism might provide the response of the tangential section of the spiral bay or pocket beach as it orients itself into the dominant direction of wave approach. Wave length is an important parameter in wave diffraction and wave refraction both of which are important mechanisms in breakwater wave attenuation and pocket beach configurations.

Bodge (1998) offers the 1/3 rule for the relationship of breakwater gap (Gb) to bay indentation (Mb) or the maximum offset of the embayed beach from a line connecting adjacent breakwaters. Bodge (2003) provides formulae to assist in developing this ratio and notes that it is a combination of the static equilibrium bay (SEB) model and his research to define mean low water around an embayed coast.

The Coastal Engineering Manual (CEM, 2000) defines the minimum beach (Bm) width as Ymin, the minimum

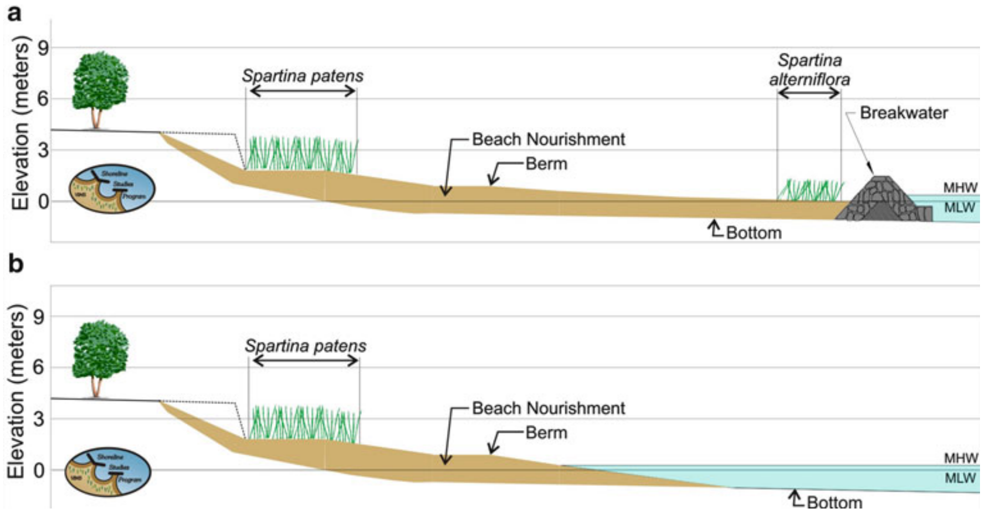
Headland Breakwaters, Figure 1 An aerial photo showing a headland breakwater project on the Potomac River within the Chesapeake Bay estuarine system soon after installation. *Inset*: A ground photo showing the extensive vegetation planting along the backshore.



Headland Breakwaters, Figure 2 Graphic description of parameters within a headland breakwater system.

dry beach width as the horizontal distance between the mean high water (MHW) shoreline and the landward boundary or base reference line. The MHW shoreline is employed because it commonly is the land/water boundary shoreline on maps, is more readily identified from aerial photos, and is a more conservative, minimum width (and volume) for shore protection. It is the minimum dry beach width required to protect the foredune, cliff, structure or vegetation under normal storm conditions. According to the CEM, the beach does the work and its resilience and recovery are critical for long-term shore protection.

Silvester and Hsu (1993) define *dynamic equilibrium* as sand transport through an embayed coast so long as the upcoast supply of sand remains constant. If the sand-supply is reduced over a reasonable length of time, the



Headland Breakwaters, Figure 3 Typical cross-section (**a**) across the beach and breakwater unit and (**b**) across a mid-bay beach.

bay will become more indented or will recede in the curved portion. Should the supply cease altogether, the waterline will erode back to a limiting shape which is termed *static equilibrium*. For coasts with predominate wave climates, this limiting shape is predictable using Static Equilibrium Model (SEB) formula. Varying wave conditions and sand supply are the norm in Chesapeake Bay, as in most estuaries, and must be accounted for in the design process.

Since the first Chesapeake Bay headland breakwater installation in 1985, creating shore protection using pocket beaches while reducing the amount of rock (i.e., breakwater length) per length of shoreline has been researched. There are numerous headland breakwater projects built in different coastal settings for shore protection and public beach stability (Hardaway and Gunn, 2002). Stability of the bay beach is critical so that future nourishment is minimized. However, rock costs for breakwater units also are significant, and it is a balance of these and project goals that make each site different. The beach is the primary component of any given headland breakwater system and the source of material will dictate costs and ultimately the design. Sand that can be obtained directly from an adjacent sandy bank will cost significantly less than sand that has to be trucked to the site.

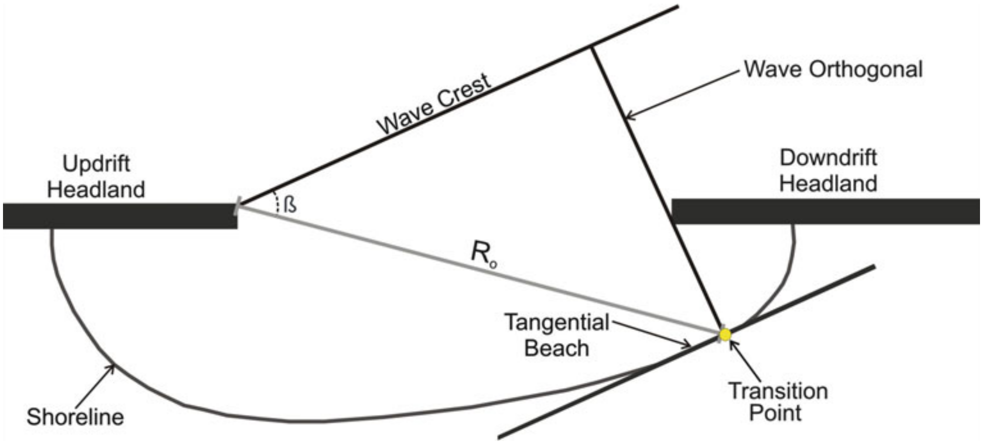
Beach berms occur on "natural" Chesapeake Bay beaches and typically reside about 0.3–0.6 m above MHW. The larger the fetch at the site, the higher the beach berm is relative to MHW due, in part, to increased wave runup. Since a stable pocket beach is the goal of the headland breakwater projects, it makes sense to build the beach berm into the project. Empirical evidence can be found on existing beaches whether natural, man-induced (i.e., jetties) or man-made (i.e., groins or breakwaters). Protective beaches also may have a storm berm that is 0.3–0.6 m above the beach berm and 1.5–4.6 m landward. The berms also provide the planting zones for upper beach (*Spartina patens)* and dune grasses (*Ammophila)* (Figure 3a, b). Often *Spartina alterniflora* can be established on the flanks of a tombolo in the lee of a breakwater unit between mean tide and mean higher-high water.

Another important design consideration is how the system interfaces with adjacent shorelines. Headland breakwaters can have a significant impact on littoral processes, and those impacts need to be assessed early in the design process. Some methods range from placing shorter, low broad structures at the "downdrift" boundary to adding more fill as a feeder beach (Hardaway et al., 1993).

Care must be taken in defining the downdrift shore. This is important because a bimodal wave climate may exist when storm wave conditions are contrary to the seasonal or annual wave field. The downdrift is more easily defined with a more unidirectional wave field. Bimodal and unidirectional conditions can be related to the shoreline setting or geomorphology and the location of the project on a coastal headland, embayment, or a relatively straight shore (Hardaway and Gunn, 2002).

Equilibrium bays

The design and performance of pocket beach shorelines have been the topic of research for many years. The SEB model is the result of years of research by Hsu et al. (1989) and Silvester and Hsu (1993) and by practical application by Hardaway and Gunn (1991, 2002) and Hardaway et al. (1993, 1995). The SEB model was developed for open ocean coasts and relatively long bays



Headland Breakwaters, Figure 4 Static equilibrium bay model parameter description (After Silvester and Hsu, 1993).

between large headlands. Relating this research to smaller estuarine systems involves understanding the goals of the project, how far offshore the breakwater units can or should be placed, how long the breakwater units and how wide the gaps should be, and how much beach fill is required. Also important is relating the geomorphic setting of the site to the wave climate. The shore direction of face, fetch, upland height, and shore configuration are important site parameters that influence the design depending on whether a unidirectional or bimodal wave climate is experienced at the site. Generally, in Chesapeake Bay, the waves are short, and the systems are scaled down.

The main component in SEB modeling is the transition position or the point of the extension of Ro (Figure 4). Ro and the tangential section of the bay describe the net direction of wave approach within the bay. They change with regard to the various directions of the incoming waves. Determining if a site is unidirectional or bidirectional is an important design component. Figure 4 shows how the wave orthogonal strikes the downdrift headland breakwater unit and continues to a point on the bay beach shoreline that defines the terminus of Ro. This takes into account the downdrift diffraction point which causes the shoreline to sit back in a small spiral. A reversal of wave climate from the other quadrant would cause the small spiral to increase until it became the main spiral section of the crenulated embayment and a countercurrent effect would occur on the updrift side as it becomes the downdrift side.

Conclusions

The Static Equilibrium Bay (SEB) model of Silvester and Hsu has shown its utility in defining the pocket or embayed beach planform between headland breakwaters. Using bay plots for varying wind/wave conditions and water levels help define the limits of shoreline change for each scenario, particularly conditions other than true unidirectional waves.

Generally, projects located in bimodal wind/wave settings should allow for what can be called omnidirectional wave attack at varying water levels. The breakwater gap (Gb) may have to be reduced relative to both breakwater length (Lb) and pocket beach indentation (Mb) so that major shifts in the beach planform will adjust within the embayment. On sites with a definite unidirectional wind/wave approach, the breakwater gap (Gb) can be opened relative to Lb and Mb. Some ratios of Mb:Gb are as high as 1:2.5, and the tangential feature of the pocket beach does not change significantly alongshore. The sand volume, i.e., the protective beach, required to be placed in headland breakwater systems is determined by the breakwater system dimensions that fall within the boundaries of the aforementioned parameter relationships (Hardaway and Gunn, 2002).

The parameter relationships are offered as a guide for breakwater design along fetch and depth limited shorelines like the Chesapeake Bay. The goal of these headland breakwater systems is to not only provide long-term shore protection but also create a stable coastal profile of beach, backshore, and low dunes that provide wetland habitat and easy access to the waters of Chesapeake Bay. Providing stable pocket beaches for long-term shore protection can be done cost-effectively. The procedures used over the years to evaluate and design headland breakwaters have been, in retrospect, effective. These installations provide a database of successful estuarine headland breakwater installations, some of which are over 20 years old. This database will continue to be used to verify and compare parameters for headland systems in the future as sites continue to mature.

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Cross-references

Coastal Erosion Control
Shore Protection
Tombolo

HEAVY MINERALS

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Definition

Heavy minerals refer to minerals with density >2.9 g/cm^3.

Description

In addition to their use as sediment source (provenance) indicators, heavy minerals (garnet, magnetite, ilmenite, tourmaline, zircon, hornblende, etc.) provide important information about hydrodynamic conditions responsible for erosion and deposition of estuarine sediments (Komar, 2007). Increases in transport energy due to floods, storms, or tsunamis often generate diagnostic heavy-mineral concentrations (HMCs) through a combination of selective sorting factors that result from differences between the light (quartz, feldspar, muscovite, calcite) and heavy-mineral fraction (Figure 1).

On shorter timescales, tidal flow regime and wave-generated flows may produce localized enrichment in heavy-mineral components. Estuarine bedforms, such as two- and three-dimensional ripples and megaripples, may cause preferential accumulation of denser minerals along their crests. In some regions of tidal channels, estuarine beaches, and adjacent dune fields, prolonged selective sorting may result in economically viable concentrations (placers). HMCs containing magnetite provide distinct magnetic susceptibility signatures in estuarine sequences making them valuable event indicators. When analyzing siliciclastic estuarine deposits, HMC thickness, concentration factor, granulometry, and spatial distribution should provide valuable information about past estuarine dynamics.

Heavy Minerals, Figure 1 Patches of magnetite and garnet sand (*center of photo*) along the intertidal shoreline of Delaware River Estuary, Pennsylvania.

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Cross-references

Estuarine Sediment Composition
Trace Metals in Estuaries

HERBIVOROUS GRAZERS

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Synonyms

Grazers; Herbivores; Primary consumers

Definition

Herbivorous grazers are heterotrophic organisms that consume algae and/or plants. A wide range of species, both in terms of body size and taxonomic diversity, are herbivorous grazers.

Summary

Herbivorous grazers occupy the trophic niche immediately above primary producers in food webs. They are commonly found in estuarine habitats and consume estuarine plants (including seagrasses, salt marsh plants, and mangroves) and/or algae (including phytoplankton, benthic microalgae, and benthic or drifting macroalgae). In estuarine systems, herbivorous grazers can be either marine or terrestrial in origin, due to the proximity of the two habitats (Denno et al., 2002; Coverdale et al., 2012).

Estuarine herbivores are frequently grouped into three general categories based upon their body size: <0.1 cm micrograzers (e.g., amoeboids, ciliates), 0.1–2 cm mesograzers (e.g., amphipods, copepods, crabs, small gastropods, insects), and >2 cm macrograzers (e.g., sea urchins, fish, birds, mammals). Herbivorous grazers can also be grouped together on the basis of their estuarine location or origin (e.g., benthic vs. pelagic vs. terrestrial). Thus, phytoplankton are consumed by zooplankton and benthic suspension feeders, benthic microalgae are consumed by benthic deposit feeders and mobile benthic invertebrate grazers, and macroalgae and plants are consumed by mobile benthic or terrestrial invertebrates, birds, and fish. Depending on the relative sizes of herbivorous grazers and their food, they may consume either entire organisms (i.e., microalgae) or parts of organisms (i.e., macroalgae, plants).

Some herbivorous grazers can also utilize macroalgae and/or plants as a habitat (Dijkstra et al., 2012), which also can provide protection from predators and/or be used as a substrate on which to lay eggs (Balouskus and Targett, 2012).

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Cross-references

Food Web/Trophic Dynamics
Heterotrophic
Phytoplankton
Zooplankton

HERBIVORY

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Synonyms

Grazing

Definition

Herbivory is a type of exploitation in which heterotrophs consume autotroph (algae and plants) biomass.

Summary

Herbivory is common and widespread in estuarine systems, and numerous invertebrate and vertebrate taxa can act as herbivores in these habitats (see *Herbivorous Grazers*).

Estuaries worldwide vary substantially in herbivore densities and consumption rates, leading to corresponding variation in the trophic impact and ecosystem-level importance of herbivory. Herbivory can shape the abundance, distribution, and species composition of primary producers. Herbivores in low-nutrient systems can serve as a "biological control" for algal blooms, for example, but may have little impact on algal biomass under other conditions (Hauxwell et al., 1998; Worm and Lotze, 2006). In many estuarine systems, primary producer biomass is mainly degraded via detrital pathways, with herbivory having a much more limited role (e.g., Conover, 2011; Guidone et al., 2012).

In systems where herbivory plays an important role, primary producers frequently utilize morphological adaptations and/or produce secondary metabolites as defenses against consumers. Morphological adaptations include spines, scales, or thick and tough outer tissue layers, while secondary metabolites can make plants and algae distasteful or toxic to grazers (Hay et al., 1994; Van Alstyne et al., 2007).

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Cross-references

Ecological Niche
Food Web/Trophic Dynamics
Herbivorous Grazers

HETEROTROPHIC

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Definition

Heterotrophic organisms use organic carbon compounds for their carbon source (Okafar, 2011). The corresponding pathways of carbon metabolism are also referred to as heterotrophic.

Etymology: from Greek ἕτερος (other, different) and Greek τροφή (nourishment).

To meet their cellular demand of carbon, heterotrophic organisms rely on organic carbon compounds that originate from other organisms. Examples include animals, fungi, numerous bacterial and archaeal species, and parasitic plants. The latter, however, have not been detected in aquatic environments (Rubiales and Heide-Jørgensen, 2011). In addition, many autotrophic organisms can operate heterotrophic metabolic pathways. Heterotrophic bacteria and zooplankton are important players in the microbial loop of aquatic food webs (Barber, 2007; Fenchel, 2008).

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Cross-references

Autotrophic
Detritus Food Webs
Microbial Degradation

I

ICHTHYOFAUNA

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Synonyms

Fish assemblages

Definition

Ichthyofauna refers to assemblages of fish in a waterbody or zoogeographic region. More broadly defined, ichthyofauna refers to the aggregate of fishes inhabiting the earth at any given period of time (The Great Soviet Encyclopedia, 2010; Collins English Dictionary, 2013).

Characteristics

The fishes are the most numerous and diverse of the major vertebrate groups, exhibiting great diversity in their morphology, biology, and habitat use. They comprise slightly more than one-half of the total number of 54,711 recognized living vertebrate species; there are valid descriptions of an estimated 27,977 species of fish (Nelson, 2006). Many fishes are recreationally and commercially important and support the economy of many nations. Of the living species of fish, ~58 % are marine, 41 % freshwater, and 1 % migrate between freshwater and saltwater (Moyle and Cech, 2000). Although freshwater covers only ~1 % of the earth's surface, ~40 % of all fishes live in freshwater. Most marine species of fish (77.5 %) live in coastal and littoral waters (Wootton, 1992). The abundance and distribution of ichthyofauna are affected by both biotic factors (e.g., schooling behavior, habitat selection, and interrelationships with other organisms) and physicochemical factors (Krebs, 1994). Thus, the ichthyofaunal species found locally in waterbodies depend on a wide range of environmental and biotic conditions. Distinct latitude patterns of ichthyofauna are also apparent, with the greatest diversity of species found in tropical regions and decreasing diversity observed toward the poles.

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Cross-references

Anadromous
Fish Assemblages
Ichthyoplankton

M.J. Kennish (ed.), *Encyclopedia of Estuaries*, DOI 10.1007/978-94-017-8801-4,

ICHTHYOPLANKTON

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Synonyms

Fish eggs; Fish larvae

Definition

Ichthyoplankton refers to planktonic portion of the life history of fishes (fish eggs and fish larvae).

Description

The planktonic portion of the life history of fishes typically begins at hatching from pelagic or benthic eggs. The duration of development of eggs and larvae is largely influenced by temperature and can last from days to months. The planktonic larval stages (preflexion, flexion, postflexion of notochord) of fishes that occur in estuaries are typically small (<20 mm but ranging up to 100 mm for the leptocephali and glass eels of anguilliform fishes). In addition, they are incompletely developed and have adaptations for living in the water column (transparent or nearly so, with developing swim bladders) at the varying salinities typical of estuaries. Many of the major transitions in the morphology, physiology, and ecology in the life history of fishes begin while they are in the plankton with such events as formation and pigmentation of the eyes, first feeding on external food, development of the gastrointestinal system, gill formation, development of ability to osmoregulate, fin ray and scale formation, and pigmentation. Less is known about the behavior of fishes in the plankton, especially in situ, because of their relative rarity. Most of the emphasis that has occurred has been on the study of feeding and schooling (both transitory and permanent) with less emphasis on predator–prey interactions and at settlement. Pelagic species remain in the water column after metamorphosis. Benthic species leave the plankton late or at the end of metamorphosis followed by settlement to juvenile habitats. The numerous transitions that occur while in the plankton may account for the variability in larval growth and survival and thus the variability in annual year class strength that is so common in fishes (Able and Fahay, 2010).

Distinct seasonal patterns in the occurrence of ichthyoplankton are well defined in boreal and temperate estuaries but are often less defined in tropical estuaries. The species composition can be derived from numerous sources including spawning in estuaries (residents), in freshwater (anadromous), in saltwater on their way to freshwater (catadromous) or estuarine nurseries, or combinations of the above. Resident species typically hatch at small sizes early in development while transients coming from the ocean are larger and later in development, often with behaviors, such as vertical swimming, that provide for tidal stream transport into estuaries. Many of these categories of ichthyoplankton include species of economic importance in recreational and commercial fisheries. Some dominant families in terms of species diversity and abundance include sciaenids, engraulids, gobiids, pleuronectids, and clupeids.

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Cross-references

Ichthyofauna
Zooplankton

INFAUNA

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Synonyms

Endofauna

Definition

The infauna refers to benthic animals that live in soft sediments.

Description

While the infauna lives in bottom sediments, those fauna living on the sediments are termed the epifauna. Benthic organisms can be categorized in different ways (see Gray and Elliott, 2009). For example, they can be separated according to (1) size, from microfauna to megafauna and from microflora to macroflora; (2) level on the shore, that is, whether they occupy the supratidal zone, intertidal zone, or sublittoral (or subtidal) zone; (3) movement, that is, whether they are mobile, or sessile; and (4) their position in relation to the sediment. Some investigators separate organisms based on whether they are moving over the sediment (mobile hyperbenthic fauna), are on the sediment (epibenthos – including the attached epiflora and epifauna and the mobile and sessile epifauna (some workers use the term exofauna)), or are in the sediment (infauna or, less commonly, endofauna).

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Cross-references

Epibiont
Macrofauna
Meiofauna
Soft Sediment Communities

INTERTIDAL ZONATION

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Definition

Intertidal zonation refers to the frequently observed pattern by which species replace one another along a gradient from the low to high tide lines along many of the world's coastlines.

Causes of zonation

Intertidal zones are the regions between the lowest and highest tides on a shoreline which serves as an interface between the terrestrial and marine environments. As such, organisms living in these often physically harsh habitats must survive both submersion (at high tide) and aerial exposure (at low tide) (Fig. 1). The resulting gradient whereby organisms closer to the ocean (in the low intertidal zone) are generally exposed to air for short periods and those higher on the shore can spend more time in air than in water has resulted in intertidal zone emergence as natural laboratories for examining how the physical and biotic environments affect the distribution of organisms in nature (Paine, 1994).

Early studies of zonation in the late nineteenth century focused on documenting patterns, but later work shifted to a focus on the underlying causes of zonation (Benson, 2002). Seminal work by Connell (1961) and Paine (reviewed in Paine, 1994) has led to the general paradigm that the lower (oceanward) edge of zones are set by biotic factors such as competition and predation, while the upper (shoreward) edges are driven by tolerance to environmental stress and emersion time. Because of the sensitivity of intertidal organisms to environmental stress, the study of the causes of intertidal zonation has renewed significance in the context of climate change impacts.

Intertidal Zonation, Figure 1 An example of zonation on the coast of Massachusetts, USA. Species replacements along an intertidal gradient can often result in sharp demarcations such as the transition from mussels to barnacles shown here.

While several schemata for defining zonation have been articulated (e.g., Doty, 1946), zones are typically defined either by their height above the low tide line (low, middle, upper, and splash zones) or by the organisms that dominate the zone (e.g., the "mussel zone" in many mid-intertidal regions).

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Cross-references

Coastal Wetlands
Littoral Zone
Tidal Flat

INTRODUCED SPECIES

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Synonyms

Alien species; Exotic species; Non-native species

Definition

An introduced species is one that arrives in an ecosystem, where it did not occur naturally, with intentional or accidental human assistance.

Biological introductions and invasions

Introductions of non-native, or exotic, species to estuaries or other new ecosystems are the direct or indirect results of human activities. These introductions do not necessarily cause economic or environmental harm, and social and economic factors are often as critical as biological factors in the introduction process (Sakai et al., 2001). However, when a species is proposed for introduction, or a recent introduction is detected, invasion science suggests cause for concern (Simberloff et al., 2013).

Today, it is accepted that the ecological and evolutionary consequences of invasions are greater than previously thought. Relatively recent studies of highly invaded coastal systems are changing our understanding of the degree to which invaders have altered natural habitats, and what is now known about the ecological and evolutionary changes brought about by non-native coastal species may be just a small part of a very large environmental problem (Grosholz, 2002).

The most common sources of introductions in estuaries are ballast water, aquarium trade, and aquaculture (Ruiz et al., 1997; Ruiz et al., 2000; Naylor et al., 2001).

After an introduction of an exotic species, the next step may be an invasion. In 1958, Charles Elton published the book "The Ecology of Invasions by Animals and Plants" which launched the systematic study of biological invasions. Invasion ecology has grown to become an important multidisciplinary subfield of ecology with growing links to many other disciplines (Richardson and Py, 2008).

Conclusions

Biotic changes in estuarine ecosystems via species invasions and extinctions caused by human activities have altered ecosystem goods and services in many well-documented cases, and several of these changes are difficult, expensive, or impossible to reverse or fix with technological solutions (Hooper et al., 2005). Invasion research has shown that the scope and complexity of consequences greatly exceed earlier perceptions (Simberloff et al., 2013). Half a century after its publication, Charles Elton's book on invasions remains influential, but massive changes in the status of invasions and other environmental issues worldwide, together with advances in technology, are reshaping the rules and priorities of invasion ecology (Richardson and Py, 2008).

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Cross-references

Exotic Species
Invasive Species

INVASIVE SPECIES

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Synonyms

Alien species; Exotics; Nonindigenous species; Nonnatives

Definition

An invasive species is an introduced species (also called "nonindigenous" or "nonnative") that adversely affects the new habitat ecologically or economically. Invasive species may be microbes, plants, or animals and may disrupt the new area by dominating particular habitats because of loss of natural biological controls present in its native range, such as predators or diseases.

Introduction

When nonnative species arrive in a new location, most of them either are harmless or do not survive. However, some do flourish, and the small percentage that thrive and reproduce may eventually become invasive by outcompeting, eating, or parasitizing native species. If a nonindigenous species displaces or competes with a native species, it reduces diversity. If a plant that functions as habitat or food for certain animals is replaced by a different plant, the animals that depended on it may be unable to survive.

Vectors: how they are transported

Early oceangoing ships were built of wood, and their hulls became covered with attached organisms such as barnacles and algae, which were transported to other ports around the world. Ships took on ballast in their home port in the form of rocks or sand in order to provide stability at sea, and terrestrial organisms in the ballast were dumped along with the rocks to lighten the ship when it arrived at its destination so it could enter a shallow harbor in a foreign port. Water is now used as ballast, and millions of gallons of water, along with resident organisms, are taken up at one port and released in another. Millions of planktonic organisms including larvae are contained in the ballast water, and when the water is taken up into the ship, some sediments are sucked into the ballast tanks as well. In this way whole benthic communities can be moved around the world.

In addition to ballast water, sea chests, the intake areas for ballast water, can house diverse organisms. They have grates or screens to prevent large organisms from coming in with the water, but small ones can survive and grow inside the chests. Sponges, sea anemones, worms, mussels, oysters, scallops, barnacles, crabs, sea urchins, and fish have been found in sea chests. Although antifouling paints have decreased hull fouling, many nooks and crannies including rudders, gratings, holes, and sea chests are difficult to paint and are subject to high levels of fouling. Fouling organisms can settle on anchors or chains as juveniles and grow into adults that are transported to a new region where they reproduce or drop off. Mud accumulates on anchors while on the bottom, and organisms in the mud remain on the anchor as it is hauled aboard ship. If it is not washed thoroughly, the organisms may survive in the damp mud until the anchor is dropped at the next port, and then they may invade the new habitat.

There are other important vectors for moving marine and estuarine species around the planet. Aquariums and trade in aquarium and ornamental species are another important source for species likely to invade aquatic habitats (Padilla and Williams, 2004). In addition, aquaculture is a major source for marine invasive species. Other mechanisms include seafood companies, bait shops, research and educational organizations, public aquariums, and coastal restoration projects (Weigle et al., 2005). Recently marine debris has also been found to be a way to transport species around the world. Almost 2 years after the major Japanese earthquake and tsunami, large pieces of docks, dislodged by the earthquake, started arriving on the US West Coast complete with large populations of attached Japanese organisms.

After arrival

There is usually a lag period of several years between the arrival and establishment of an exotic species and its rapid population growth. Scientists think that stressed or disturbed environments are more susceptible to invasion, so that natural communities that have been changed by human impact may be more vulnerable. When native species are reduced in numbers or impaired, it is easier for new species to succeed, and the relative scarcity of certain functional groups (e.g., filter feeders) will facilitate invasions by other species of those groups. If there are unused resources such as nutrients or unused habitats (e.g., hard surfaces), an area will be more susceptible to invasion by species that use those resources. For example, European brackish water seas (Baltic Sea, Black Sea, Caspian Sea) are subject to intense invasion. In these seas, natives have rather low species richness because salinity is the most important limiting factor. However, this salinity is very suitable to many invaders, which are abundant in these brackish waters (Paavola et al., 2005). Nehring (2006) studied why estuaries seem to be invaded more than other ecosystems and concluded that the combination of brackish water with unsaturated ecological niches plus intensive international ship traffic produces the highest potential invasion rate for aquatic systems. Estuaries are also subjected to invasions both from the ocean (mainly shipping) and from inland waters (e.g., canals).

We cannot predict when or where a species will become a problem, a phenomenon that has been called “ecological roulette” (Carlton and Geller, 1993). Why does a species arrive at a particular time? Why not last year or 10 years ago? Sometimes the explanation is a change in the donor region such as an increase in population or a change in the recipient region such as diminished water quality due to a new source of pollution. The possibility of invasion is increased by more and faster transport between regions that are climatically similar and the use of bigger ships carrying more ballast water. The number of individuals introduced at one time and the frequency with which they arrive, called “propagule pressure,” play an important role. Sometimes sheer numbers can make the difference between success and failure of an introduced species.

Degree of the problem

In the waters of the United States, it was estimated a few years ago that there are over 500 species of introduced organisms, including over 140 in the Great Lakes, over 200 in Chesapeake Bay, and over 240 in San Francisco Bay. In San Francisco Bay alone, a new species is estimated to arrive every 14 weeks (Cohen and Carlton, 1998). More than half of the area’s fish and most of its benthic organisms are not native. Hawaii is another region with high levels of invasion, with 73 marine invasive species, 42 % of which are considered harmful.

Effects

Invasive species can cause major economic problems. The invasion of zebra mussels in the Great Lakes in the late 1980s had a major economic impact because they clogged up intake pipes for the water systems in the Great Lakes. It is estimated that invasive species cost the world’s economy hundreds of billions of dollars annually; the cost to the United States is estimated at 128 billion dollars each year. Much of this cost is due to impacts of invasive species on fisheries, boating, and coastal recreation, as well as to the expense of controlling and/or attempting to eradicate them (Pimentel et al., 2000).

Invaders can be major sources of stress in estuaries (Ruiz et al., 1999), although many nonindigenous species do not appear to have major impacts. One type of invader that can have major impacts is termed an “ecological engineer.” These species modify their habitat, for example, by changing the substrate, changing light penetration, digging burrows, or elevating the marsh surface

(Wallentinus and Nyberg, 2003). Changing the substrate makes the habitat unsuitable for many native species and more suitable for a new group of species; reducing light penetration harms rooted sea grasses that need to photosynthesize. When the marsh surface becomes more elevated, the amount of time that it is underwater during the tidal cycle will be reduced. Organisms that dig burrows at the marsh edge can contribute to erosion and loss of marsh area.

Many invaders become larger in their new location than in their native range (Grosholz and Ruiz, 2003), perhaps due to the presence of fewer predators at the new location. This is referred to as the "enemy release hypothesis." Parasites are often lost during the transition to a new location. For example, if a species arrives as planktonic larva, it will not have brought along the parasites that are present in the adult animal. Parasites often have complicated life cycles that require more than one host, and if all the hosts are not present in the new location, those parasites cannot survive. Reduced predators and parasites can contribute to the increased size and success in the new location.

Examples of some major troublemakers

The common reed: Phragmites australis

East Coast brackish marshes have been invaded by a new European strain of the common reed, *Phragmites australis* (Saltonstall, 2002). This species has been in North America for thousands of years, but until recent decades it remained a minor part of the high marsh and freshwater marsh communities until the arrival of the new strain. This particular strain, which may have arrived from Europe in dry ballast in the hold of a ship during the nineteenth century, produces more shoots and grows faster than the native variety, outcompeting and reducing the diversity of native marsh plants. It expands rapidly using rhizomes which can grow more than 10 m in a growing season, and it is more tolerant of salt and low oxygen than the native variety. It also spreads easily in low-salinity areas where tidal flow is restricted, and it is more dense and taller than native plants, crowding them and blocking their exposure to the sun. It produces dense litter layers that sit on the marsh surface and inhibit germination and growth of other species. It thrives on high levels of nitrogen found in estuaries receiving sewage plant effluent or farm runoff, giving it yet another advantage over native plants (Figure 1).

Managers have established removal policies costing millions of dollars. In many areas where it was removed, the reed has reinvaded, and at the same time research has found that its effects on the marsh are not all negative. While it does reduce plant diversity, many animals do not seem to be affected by its takeover of the marsh. On the negative side, it has reduced value as a nursery habitat for killifish; larval and juvenile killifish are much less abundant in *Phragmites* marshes because these smooth out the marsh surface, removing the small puddles and depressions that the larvae and juveniles live in at low tide (Able and Hagen, 2000). On the positive side, invertebrates such as fiddler crabs, ribbed mussels, grass shrimp, and benthic infauna seem to thrive regardless of which species of marsh plant is dominant (Weis and Weis, 2003). Birds are mixed – some prefer *Spartina*, but some prefer *Phragmites* (Benoit and Askins, 1999). The detritus produced by decaying leaves of the reed provides nutrition comparable to that of cordgrass, and it enters estuarine food webs the same way as cordgrass detritus.

Another way that the common reed changes the marsh is by increasing the rate at which sediment is trapped, filling in creeks and tidal channels, increasing the elevation of the marsh (Rooth et al., 2003). The overall smoothing and filling of the marsh surface reduces its variability and reduces habitat available for animals. Ditches and creeks can become filled, reducing tidal exchange with the high marsh and limiting access for aquatic organisms to go up on the marsh surface. In some areas, the marsh may eventually become so elevated that it will be higher than the upper tide level and will dry out. Ironically, given the forecasted rate of sea level rise, it may be that only *Phragmites* marshes will be able to keep pace and survive.

The reed can sequester pollutants more effectively than native cordgrass. It absorbs more nitrogen, which reduces effects of the nitrogen on the rest of the marsh/estuary community. It retains more metal pollutants belowground in its roots, while *Spartina* puts more into its leaves and excretes them back into the water. Keeping the contaminants belowground, as *Phragmites* does, is beneficial to the ecosystem since the contaminants are not available to other species, while contaminants that are excreted back into the water can then be taken up by other species (Windham et al., 2001). *Phragmites* also appears to be more tolerant of contaminants than other marsh grass species (Valega et al., 2008).

Because of the controversy over this species, a conference entitled "*Phragmites australis*: A sheep in wolf's clothing?" was held to focus on and review all the aspects of the invasion and effects of this plant. The plant's ecology was examined with regard to how it affects habitat quality in marshes. The conclusion was that it is neither a villain nor a hero. Given the scientific findings about this plant, managers should reconsider their automatic and expensive extermination programs and replace them with well-thought-out goals based on site-specific findings.

Cordgrass: Spartina alterniflora

It is ironic that while *Spartina alterniflora* is valued in East Coast marshes, in West Coast marshes it is considered a noxious invasive species because it converts valued mudflat areas into marsh. Mudflats, rather than marshes, are considered the most biologically productive estuarine habitats because they are teeming with invertebrates of all kinds. Marine worms, crustaceans, and mollusks support the hundreds of thousands of shorebirds in the estuary. When Eastern cordgrass dominates marshes above the mudflats, it displaces native cordgrass and other native

Invasive Species, Figure 1 Photo courtesy of P. Weis.

plants, creating a single ecological zone that eliminates the transitional areas native species need. It reduces the species richness of benthic organisms and modifies benthic communities (Neira et al., 2005). It has displaced native flora, changed sedimentation, decreased invertebrate and algal populations, and eliminated foraging sites used by marsh birds and shorebirds. Levin et al. (2006) concluded there had been a "trophic shift" in the ecosystem in response to the *Spartina* invasion, involving loss of key trophic support for fishes and migratory birds by shifting dominance to species not widely consumed by species at higher trophic levels.

It is suspected that *S. alterniflora* was introduced into Willapa Bay in the late 1800s or early 1900s as packing material for oyster shipments from the East Coast. From the middle of the twentieth century, the plant spread rapidly throughout Willapa Bay. In the 1970s, the US Army Corps of Engineers deliberately introduced it to stabilize flood control levees on Alameda Island in San Francisco Bay because it grows much faster than the native cordgrass (*S. foliosa*). Unfortunately, *S. alterniflora* both outcompetes it and hybridizes with *S. foliosa* (Daehler and Strong, 1997) and the hybrid is particularly invasive, choking off small creeks in the marshes that are used by the endangered California clapper rails and covering mudflats that provide food for the rails and other shorebirds. Authorities are concerned that cordgrass is likely to choke tidal creeks, dominate newly restored marshes, impair thousands of acres of shorebird habitat, and eventually cause extinction of the native Pacific cordgrass. By 2005, about 10 % of the acres of tidal flats in the San Francisco Bay estuary had been invaded by the hybrid.

In Washington State, the loss of mudflat habitat harms marine species such as the juvenile chum salmon, Dungeness crab, and English sole that rely on these habitats as food sources. About one quarter of the total foraging habitat of the Dungeness crabs has been lost because the rigid structure of the shoots seems to reduce the ability of the crabs to access their prey (Feist and Simenstad, 2000). The sediments around the *Spartina* hybrid have also reduced communities of invertebrates, and commercial oyster production has been threatened because culture beds are being invaded. At the Willapa National Wildlife Refuge in Washington, *Spartina* has displaced habitat for wintering and breeding migratory aquatic birds.

Costly efforts to spray and excavate are underway in San Francisco Bay and Puget Sound. Workers are spraying many acres with the herbicide Habitat® which kills the grass with minimal effects on animals because it works by selectively interfering with plant biochemistry. Biological control is being investigated using the plant hopper *Prokelisia marginata* (Grevstad et al., 2003).

Researchers in China are concerned about how the invasive *Spartina alterniflora* is threatening their valuable *Phragmites australis* marshes. It is highly invasive on the Chinese coast and is causing concern due to its rapid expansion and damage to native ecosystems. Both tall and dwarf forms of the species are in China, and the tall form with its strongly invasive ability has expanded widely and is replacing native *Phragmites* in some sites. Several methods for *Spartina* control in China, such as harvesting, herbicide application, and freshwater irrigation, have been developed, but more research is needed to verify their effectiveness (An et al., 2007).

Invasive Species, Figure 2 Photo courtesy of NOAA.

What people on the East Coast call productive vegetation is considered an invasive weed on the West Coast and China. On the East Coast, *Spartina alterniflora* is planted and *Phragmites* is eradicated, while in Europe and China they value *Phragmites* and are concerned about it dying back. It does make you wonder if a certain amount of xenophobia is involved in our responses to nonindigenous species.

Green algae: Caulerpa

A strain of *Caulerpa taxifolia* developed for use as an aquarium plant was inadvertently released into the Mediterranean Sea from the Monaco Aquarium in the 1980s. It initially occupied only a few square feet, but was not removed and subsequently grew and spread so vigorously that it practically took over the Mediterranean (Meinesz et al., 1993). It can regenerate from fragments, so physical removal can make the problem worse. In areas where it grows, hardly any fishes and no other algae or invertebrates are found. To make matters worse, it contains toxic chemicals that keep grazing animals from eating it and inhibit the growth of sea grasses and other plants (Piazzi et al., 2001). *Caulerpa* rapidly overgrew native macroalgae so species cover, numbers, and diversity greatly decreased. During a period when cover and biomass of *Caulerpa* diminished, the macroalgal community did not return to initial conditions. Instead, the structural changes increased in the following season, indicating the importance of the invasion. It grows unchecked and is considered one of the most harmful marine invasive species (Figure 2).

Its presence was discovered in June 2000 in California in an eelgrass bed in Agua Hedionda Lagoon. Because of knowledge about its spread in the Mediterranean Sea, it was already on the US Federal Noxious Weed list. Alarm about its potential damage prompted rapid action and cooperation among governmental agencies, private groups, and nongovernmental organizations. The San Diego Regional Water Quality Control Board deemed this invasion tantamount to an oil spill and provided emergency funding and mobilized crews of divers. California passed legislation prohibiting the possession and sale of *Caulerpa*, and it was successfully eradicated (Anderson, 2005). California's integrated rapid response and financial resources resulted in effective containment and eradication of the alga (for the time being), one of the few examples of rapid, successful eradication. But despite the concerns of ecologists and conservationists, it is still possible to purchase *Caulerpa* from aquarium supply stores and through the Internet, so new releases may be inevitable. A similar response will undoubtedly be necessary in the future to eradicate other invasive species before they spread and become a big problem.

Red algae: Eucheuma (*Kappaphycus*)

Eucheuma is cultured throughout the Pacific. It is a major source of agar and carrageenan, which are used as food additives for gelling and stabilizing; small amounts are used by the pharmaceutical and cosmetic industries. It was imported by the late Maxwell Doty's research lab in Kaneohe Bay, Hawaii, from the Philippines in the 1970s, and this research helped the Philippines and other nations establish multimillion-dollar industries based on the algae.

The methods developed made *Eucheuma* the world's most widely farmed seaweed. However, it was released from open-cage experiments into Hawaiian coastal waters where it smothers and kills coral, creating devastation. It spread and smothered at least half the reefs in Kaneohe Bay (Conklin and Smith, 2005).

An effective tool to combat invasive algae in Hawaii is the Super Sucker, an underwater vacuum cleaner that scoops up about 800 lbs of the invasive algae each hour. It sucks the algae off the reef and places it on a barge above water so that noninvasive marine life can be sorted out and returned to the water. The alien algae are packed into sacks and delivered to taro farmers for use as fertilizer. Noted coral reef biologist Thomas Goreau thinks that this will give only temporary results unless the excess nutrients that promote the algal growth are removed. In a letter to Science Magazine, he wrote, "No amount of sucking them off will work when they grow right back because they are overfertilized. It is the suckers paying for this well-intentioned, but ultimately futile, effort who will be hosed unless the underlying causes of eutrophication are removed" (Goreau, 2008).

A long-term solution may depend on increasing the populations of sea urchins that eat the invasive algae (Conklin and Smith, 2005). The sea urchin population around Hawaii has declined because of excessive harvesting for their gonads, prized by sushi eaters. Researchers are learning how to propagate them so that they can place baby urchins on newly vacuumed reefs to graze on little bits of algae.

Common periwinkle snail: Littorina littorea

The common periwinkle was introduced from Europe to the East Coast of the United States but the dates are uncertain. Using molecular genetics, Blakeslee et al. (2008) estimated it was about 500 years ago. It feeds primarily on algae on rocks at the shoreline. The snail is so abundant that most people think of it as a normal part of the coastal habitat. It is found on rocky shores in the northeast, where it has become so abundant that its voracious grazing has destroyed much of the intertidal algal vegetation. It can also live in intertidal marshes and subtidal areas. It has caused a gradual transformation of the New England coast from rocks with abundant seaweed cover to the bare rocks that most people today think of as natural (Bertness, 1984). The state of New Hampshire actually passed a law protecting this edible species from out-of-state poaching. In salt marshes, the snails are eaten by diamondback terrapins and by blue crabs.

Asian clams: Potamocorbula amurensis

In the late 1980s, the Asian clam became established in San Francisco Bay, replacing the native clam, *Macoma balthica*, as the dominant benthic macroinvertebrate (Nichols et al., 1990). It is a filter feeder, and its rapid spread has reduced phytoplankton in the bay, which means less food for zooplankton, which means less food for fish. The invader is eaten by diving ducks and lives closer to the surface of the mud than *Macoma*, increasing its availability to the ducks. This clam has altered nutrient cycling and caused major ecosystem changes.

A related clam on the East Coast is *Corbicula fluminea*, which arrived in the late 1970s and is abundant in the Potomac River. When it was first discovered, scientists were worried that it would cause major problems like the zebra mussel in the Great Lakes (attaching to surfaces, blocking intake pipes for water supply systems, etc.), but it does not attach to hard surfaces since it lacks byssus threads. This clam instead settled on sandy bottom sediments, and after it arrived there were increases in submerged aquatic vegetation and fish (Phelps, 1984). It also serves as food for birds and muskrats, and after its arrival the water quality and clarity actually improved, possibly due in part to its filter feeding. The improvement in water quality may also have been due to improved sewage treatment and a ban on phosphorus in laundry detergents, but at the very least, this outcome means that not all invasions are ecological disasters.

Green crab: Carcinus maenas

Called "green crab" in the United States and "shore crab" in Europe, this species is native to the Atlantic coasts of Europe and Northern Africa, found on protected rocky shores, pebbly beaches, mudflats, and tidal marshes. It thrives in a wide range of salinity and temperature and has invaded numerous coastal areas, including South Africa, Australia, and both coasts of North America. Although it is called a green crab, large individuals (about 3 in across) may be brownish red. Its larvae can survive in the plankton for over 2 months, dispersing many miles up and down the coast. After the larvae develop in the sea, they are swept into coastal waters and estuaries by tides and currents where they molt and settle out as juveniles in the intertidal zone. If conditions are suitable, they will survive and reproduce, establishing new populations and extending its range (Figure 3).

There have been two major invasions of America, first in the mid-1800s when they reached the Atlantic coast, probably in crevices of fouled ship hulls. They found suitable habitat in coastal bays from New Jersey to Cape Cod. In the early 1900s, they spread north to Nova Scotia, and their arrival in Maine in the 1950s coincided with dramatic declines in the soft clam fishery. The second major invasion was detected in 1989 in San Francisco Bay where they probably arrived as larvae in ballast water of commercial ships (Grosholz and Ruiz 1996). It is also possible that they were hanging onto the fouling on ships or could have been in with the sea grass and kelp used for packing and shipping lobsters and bait worms to the West Coast.

It is an effective forager, able to open bivalves more quickly than most other crabs, which makes it a strong competitor for the food sources of native crabs, fishes, and birds and a threat to shellfisheries. It feeds on many organisms, including clams, oysters, mussels, marine

Invasive Species, Figure 3 Photo courtesy of P. Weis.

worms, and small crustaceans. It also appears to learn and remember better than native blue crabs (Roudez et al., 2007).

It threatens Dungeness crab, oyster, and clam fisheries and aquaculture in the Pacific Northwest as it moves up the West Coast (Jamieson et al., 1998). On the East Coast, the snails and mussels that have been living with it for over a century have developed thicker shells as a defense, making it harder to crush than shellfish that have not been exposed to green crabs (Leonard et al., 1999). The green crabs in North America have fewer parasites (Torchin et al., 2002) and grow considerably larger than they do back home in their native European waters, traits that may be contributing to their success.

Chinese mitten crabs: Eriocheir sinensis

The Chinese mitten crab is a burrowing crab native to the Yellow Sea in Korea and China. It has dense patches of hairs on its claws, hence the name mitten crab. In Asia, it is a delicacy and it has been imported illegally into seafood stores in California for the Asian market, suggesting that it may have been introduced by the release of leftover crabs. It could also have been accidentally released in ballast water, which seems to be what happened in the early 1900s in Germany, with the result that in the 1920s and 1930s, the crabs expanded into many northern European rivers and estuaries. Recently, the Thames River in England has had a population explosion of the crabs (Figure 4) (Herborg et al., 2005).

Invasive Species, Figure 4 Photo courtesy of Wikimedia.

They have become established on the West Coast of the United States, posing a potential threat to native invertebrates, to the ecological structure of freshwater and estuarine communities, and to some commercial fisheries. In 1992, shrimp trawlers collected the first mitten crabs in San Francisco Bay, and they have become established in the bay and have spread upstream; by 1998, they were found far up in the Sacramento River system and throughout the Delta (Rudnick et al., 2003). They may imperil threatened and endangered salmon populations due to their appetite for salmon eggs, and there is concern that they will next invade Oregon, Washington, and British Columbia. Crabs have been found in parts of Chesapeake Bay, in Delaware Bay, and in the NY/NJ region, but have not (yet) become numerous.

Adults migrate downstream to reproduce in estuaries, and after 1 or 2 months as planktonic larvae, they settle out in brackish water in late spring and migrate upstream to freshwater (Dittel and Epifanio, 2009). Juveniles primarily eat vegetation, but as they grow, they prey more on animals, particularly on worms and clams. Many animals prey on them, including bullfrogs, raccoons, river otters, wading birds, and fishes, but the predators have not been able to slow down their invasion.

Mitten crabs can walk on land and have been found on roads, parking lots, and yards and in swimming pools. Because they can leave the water and cross dry land and enter new rivers, their ability to disperse is a great concern. In tidal areas, they burrow into banks during low tides, and there is concern that this will increase erosion and make the levees and riverbanks unstable (Dittel and Epifanio, 2009).

In Europe, the primary economic impact has been the damage to commercial fishing nets and to the catch if the crabs are caught in large numbers. In San Francisco Bay, one trawler reported catching over 200 crabs in a single tow on several occasions, a time-consuming and costly diversion. Another problem in California is the impact on water diversion and on fish salvage facilities. Crabs followed the moving water into a salvage facility in the late 1990s and clogged holding tanks full of the fish that were the object of the salvage attempt (Veldhuizen and Stanish, 1999). Many fish suffocated because it took too long to separate them from the crabs, and the fish that survived were put in transport trucks, but most of them also died.

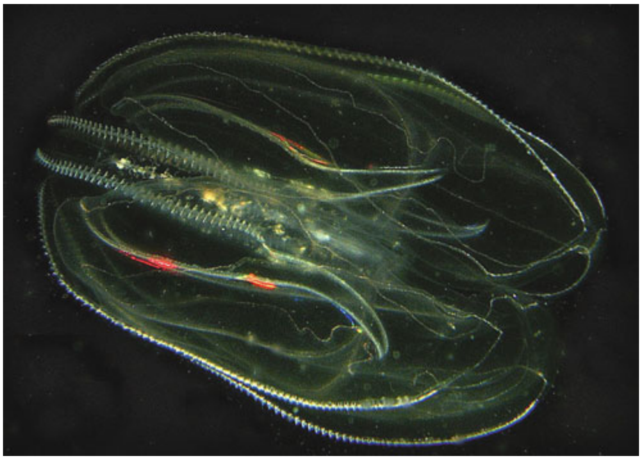
Mitten crabs are a delicacy in China, where they are not only caught but farmed in aquaculture facilities. Since they are edible, people can be encouraged to eat them to minimize potential problems. An "Invasive Species Cookbook" is available ("if you can't beat, 'em eat 'em"), and the idea of the cookbook is to increase interest in the issue of invasive species and to reduce their populations by harvesting them as food sources and finding interesting ways to eat seaweeds, crabs, clams, mussels, fish, and snails.

Asian shore crab: Hemigrapsus sanguineus
This small species was first observed in New Jersey in 1988, probably having arrived in ballast water, and it has since extended its range from Maine to North Carolina, becoming very abundant in intertidal and shallow water habitats where rock cover is available (McDermott, 1998). They have a wide diet and a long breeding season and reproduce readily in a wide range of environmental conditions. They are found in very high densities and actually seem to have displaced green crabs from some areas. Control of this species by predators seems unlikely as there is little evidence that they have major predators, supporting the "enemy release hypothesis" of invasion biology.

Comb jelly: Mnemiopsis leidyi
This comb jelly (or ctenophore) is native to the East Coast of North America. *Mnemiopsis leidyi* was introduced in the Black Sea in the 1980s, probably by ballast water. By 1989, the Black Sea population had reached 400 per m^3 of water (>10 animals/cubic foot) (Kube et al., 2007). Afterward, due to depletion of plankton, the population dropped somewhat. *M. leidyi* eats eggs and larvae of pelagic fish and caused a dramatic drop in fish populations, including the commercially important anchovy *Engraulis encrasicolus*, by competing for food and by eating the young and eggs. Another accidental invader, a larger ctenophore, *Beroe ovata*, arrived some years later. This predator ate *Mnemiopsis* and reduced its population considerably (Kideys, 2002). This is an example of a "top-down" effect, and it appears as if a fairly stable predator-prey balance has been achieved. In 1999 *Mnemiopsis* was introduced in the Caspian Sea and rapidly depleted 75 % of the zooplankton, affecting the entire ecosystem. Since then, it has spread through the Mediterranean and the northwestern Atlantic including the North Sea and the western Baltic Sea (Figure 5) (Faasse and Bayha, 2006).

Control of invasive species

Many species of nonnative invertebrates and fishes have established themselves in new locations as a result of human activities, and the rate of introduction has been accelerating, probably because of increased shipping due to larger ships and faster transoceanic voyages. It is, of course, more cost-effective to prevent invasive species from establishing themselves than it is to try to eradicate them once they have become established. Management of estuarine invaders is a difficult problem (Williams and Grosholz, 2008).



Invasive Species, Figure 5 Photo from Wikimedia.

To be useful, a plan to prevent future introductions must include understanding the pathways or vectors through which introductions take place. This awareness has led to some steps to reduce a major vector, ballast water. Ships must now exchange ballast water in the middle of the ocean; this releases organisms from the port of origin where they are unlikely to survive. The ships then take in ocean water with its planktonic organisms which are released in the destination port (Dickman and Zhang, 1999). Oceanic plankton are not likely to be able to survive for long enough to establish themselves in the variable environmental conditions in a port. This exchange is practiced by most vessels, but it is not performed in stormy seas when the release of ballast water could destabilize the vessel. These procedures can reduce concentrations of zooplankton 90 %, which should reduce the number of future invasions. However, the mud on the bottom of the ballast tanks, with its resident organisms, is not removed during ballast water exchange. Under the ballast water is a thick layer of mud with populations of marine worms that have been living there and cruising around the world for many years.

Efforts are underway to develop treatment techniques to kill the organisms in ballast water (Waite et al., 2003). These techniques could potentially be harmful to the environment if they involve toxic chemicals. Studies have tested the ability of ozone treatment to kill the organisms, and the approach appears promising.

With regard to the other major vectors, education of fishers, aquarists, aquaculturists, and the general public is the major approach to reduce or prevent future invasions. Many educational outreach programs and posters have been created.

Another approach to controlling invasive species is to evaluate those species that have become established to see if they can be contained or to attempt to eradicate them in an organized manner. Such is the case with invasive marsh plants on the East and West Coasts, where *Phragmites* and *Spartina* removal, respectively, is common.

It is rare to be able to totally eradicate a species as was done with *Caulerpa* in California.

Despite these efforts, it is likely that continued environmental changes in the twenty-first century will alter both the availability of species for transport and the degree of susceptibility to invasions. In addition to human-caused modifications in the local environment, climate change will interact with the arrival of nonnative species in new areas to modify diversity and to alter the functioning of ecosystems around the globe.

Summary and conclusions

Many species of nonnative organisms have established themselves in new estuaries as a result of human introduction, and the rate of introduction has been accelerating, probably because of increased shipping with larger ships and faster voyages. Invasions have caused extensive ecological changes in salt marshes and other coastal habitats. It is more cost-effective to prevent invasive species from establishing themselves than it is to try to eradicate them once they have become established. Invasive species threaten biodiversity, fishing, tourism, and human health. While some progress is being made, effective solutions appear to be a long way off (Bax et al., 2003).

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KARST PROCESSES AND ESTUARINE COASTLINES

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Synonyms

Coastal caves; Limestone dissolution

Definitions

Karst is a landscape created by dissolution of soluble rock, made up of caves, springs, sinkholes, sinking streams, and unusual rock sculpture.

Karren refers to the various etchings, carving, and sculpting of soluble rock by dissolution; these features are in the decimeter to meter scale.

Carbonates are sedimentary rocks consisting of calcium and/or magnesium combined with $CO_3^{=}$.

Flank margin caves are caves developed by mixing dissolution within the margin of the freshwater lens.

Sea or littoral caves are caves produced by wave energy and bioerosion on rocky coasts.

Syndepositional caves are caves produced within a soluble rock as that rock is being deposited.

A hybrid cave is a cave developed by one process, overprinted by a second process, such as a flank margin cave overprinted by littoral processes.

Introduction

Soluble rocks are common on coastlines around the world. These soluble rocks are primarily limestone ($CaCO_3$) and related rocks like dolostone ($CaMg[CO_3]_2$) because gypsum ($CaSO_4 \cdot 2H_2O$) and halite (NaCl) are too soluble and are quickly removed. In the tropics and subtropics, limestones (called carbonates to refer to all the common limestone and dolostone types) are being deposited today and tens of thousands of kilometers of coastline at low latitudes consist of these deposits (Wilson, 2012). At higher latitudes, older limestones are often present on the coast for long distances, such as along the Adriatic Sea. The soluble nature of limestones means they can dissolve to form a myriad of landforms collectively named karst: caves, springs, sinkholes, sinking streams, and unusual rock sculpture called karren (e.g., Ford and Williams, 2007; Palmer, 2007). The coastal environment creates a novel karst setting resulting from the outflow of fresh groundwater of the freshwater lens and the land mixing with salt water invading from the sea (Figure 1a) (Scheffers et al., 2012). The mixing of these two waters, even if both are initially saturated with $CaCO_3$, results in brackish water that is capable of additional dissolution (Figure 1b). The dissolution of limestone bedrock in the coastal mixing zone is accelerated compared to that done in continental interiors. The signature landform of the carbonate coastal environment is the flank margin cave, so named for its development by mixing dissolution in the discharging margin of the freshwater lens, under the flank of the enclosing landmass (Mylroie and Carew, 1990). Flank margin caves are part of the Carbonate Island Karst Model, or CIKM (Mylroie and Mylroie, 2007), developed to explain the various karst processes and phenomena that occur in carbonate coasts. Carbonate coasts develop a unique type of karren influenced by the youthfulness of the rock, boring and grazing by organisms, wave action, and mixing of sea and fresh waters; see Taboroši and Kázmér (2013) for a full and complete review of coastal karren.

Flank margin caves range in size from voids only a few meters across to extensive systems with 20 km of linear survey (Figure 2). They form very rapidly, and on many carbonate coasts fossil caves above modern sea level, with

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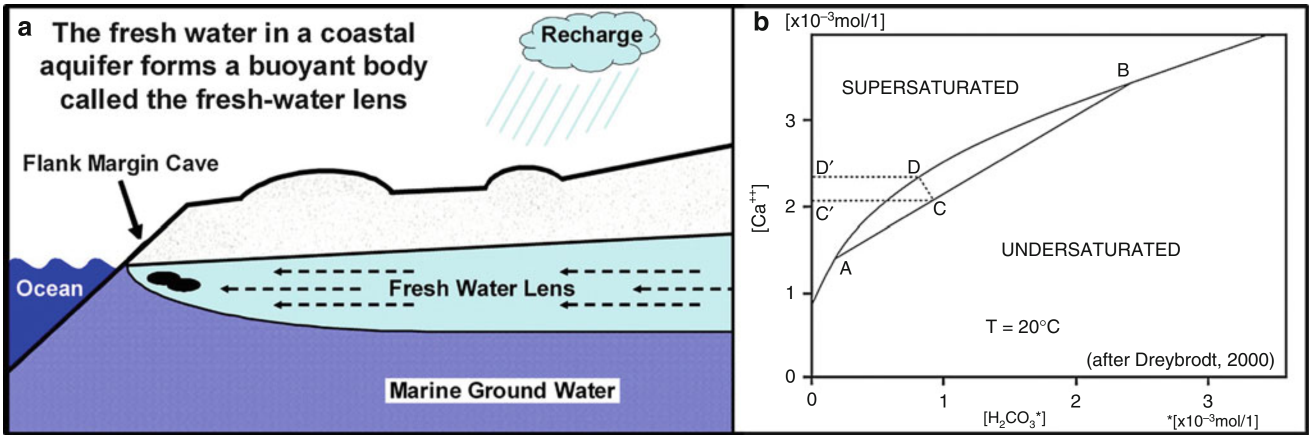
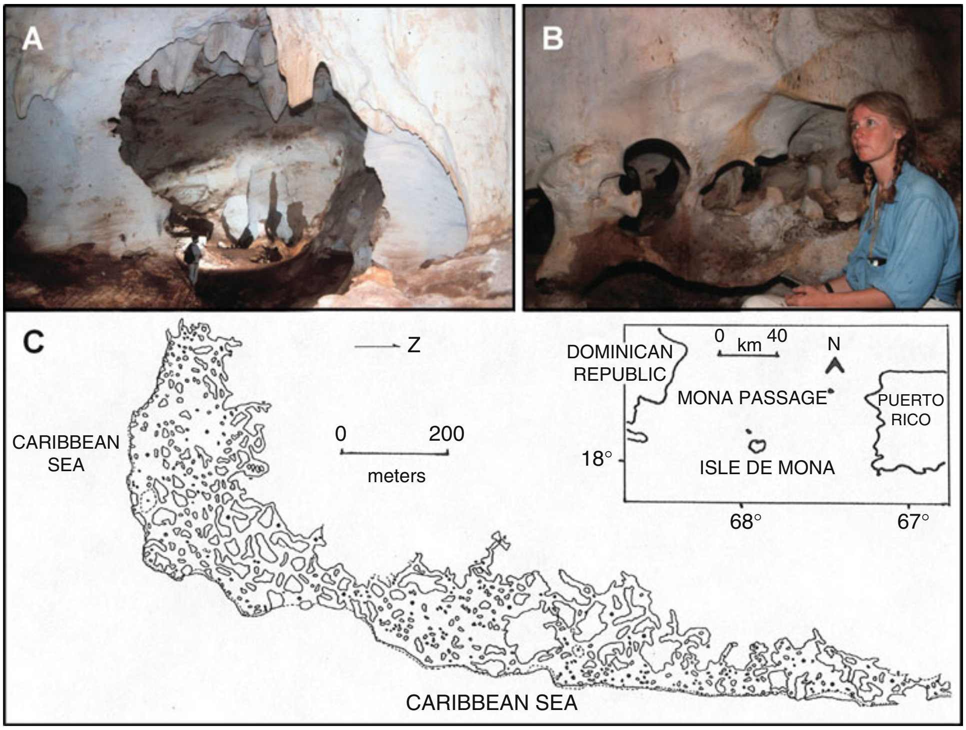
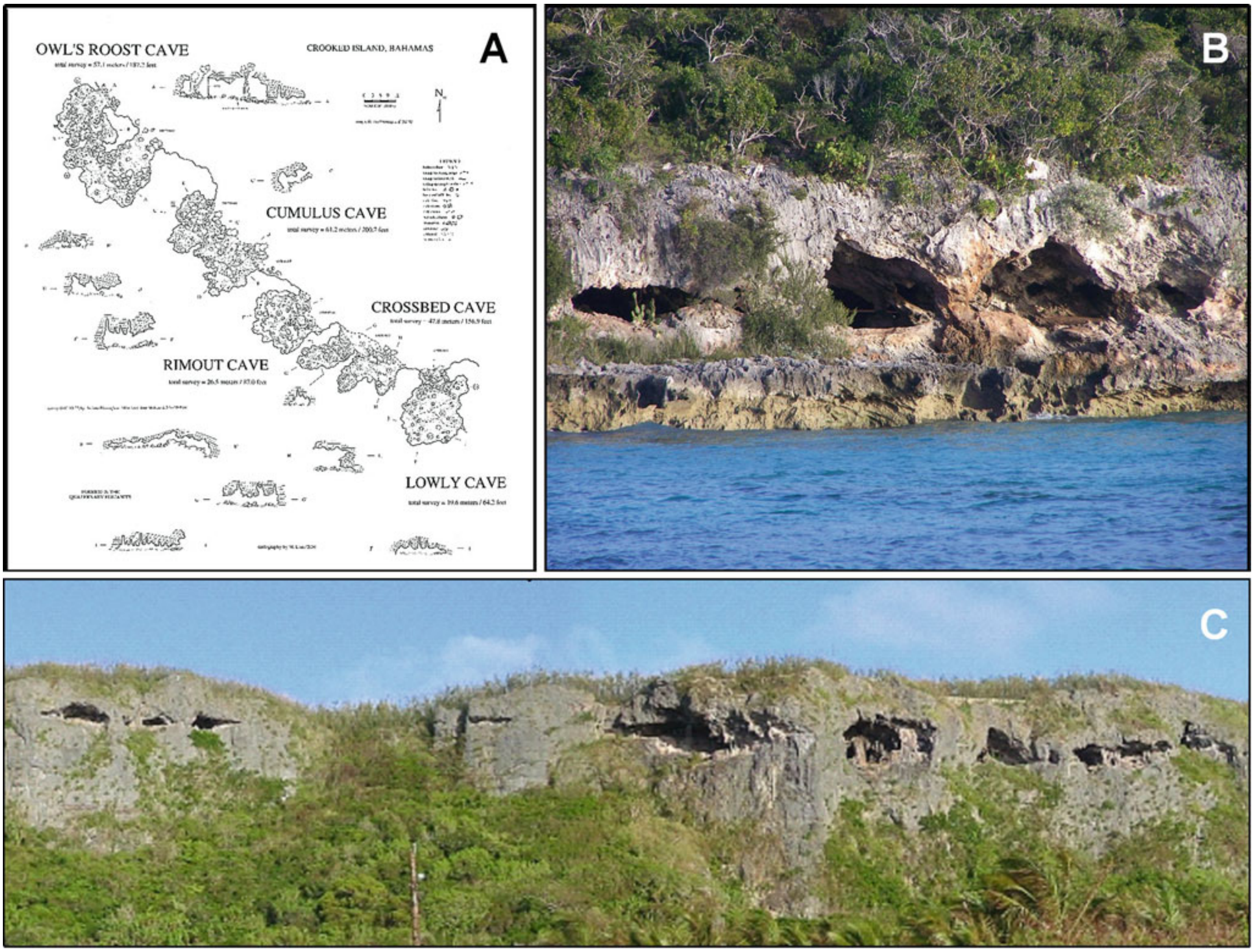


Karst Processes and Estuarine Coastlines, Figure 1 The freshwater lens and carbonate geochemistry. (**a**) Diagrammatic representation of the freshwater lens in a carbonate coast, showing the location of flank margin cave development. (**b**) Curve of $CaCO_3$ saturation as Ca^{++} vs H_2CO_3. Because the curve is convex upward, mixing of waters with different initial conditions results in the resulting mixture falling into the undersaturated zone beneath the curve (as at *C*), allowing more dissolution to occur (up to *D*), even if both waters (*A* and *B* on plot) were initially saturated with respect to $CaCO_3$.



Karst Processes and Estuarine Coastlines, Figure 2 Flank margin cave configuration. (**a**) Large passage in Hamilton's Cave, Long Island, Bahamas, a flank margin cave that developed during the last interglacial sea-level highstand (MIS 5e or Marine Isotope Substage 5e), which lasted from 124 to 115 ka (Thompson et al., 2011) and placed the freshwater lens 6 m above modern levels. (**b**) Dissolution sculpture in Harry Oakes Cave, New Providence Island, Bahamas, another flank margin cave formed during MIS 5e; such wall configuration is typical of mixing-zone dissolution. (**c**) Sistema Faro, Isla de Mona, Puerto Rico, the largest surveyed flank margin cave in the world (~19 km of survey); note how the cave passages have a ramiform pattern and wrap around the island perimeter, following the speleogenetic distal margin of the freshwater lens (cartography by M. Ohms).

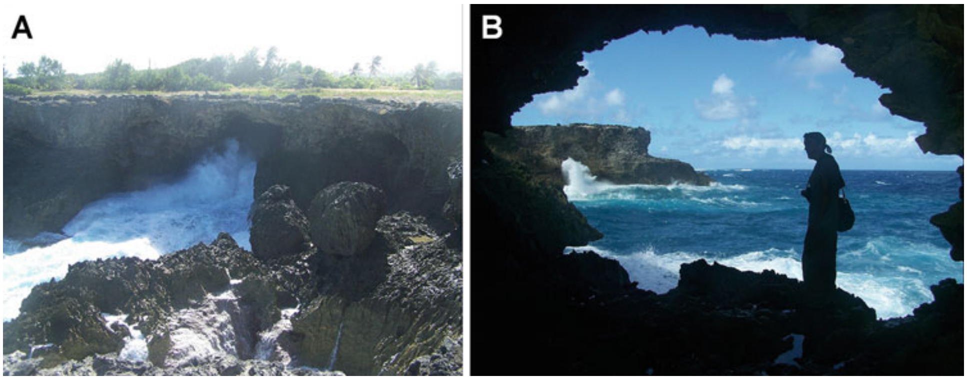


Karst Processes and Estuarine Coastlines, Figure 3 Flank margin cave "beads-on-a-string" pattern. (**a**) Map of five flank margin caves on Crooked Island, Bahamas, distributed along the side of an eolian ridge at a consistent elevation. (**b**) The beads-on-a-string cave pattern, Rum Cay, Bahamas; the caves are subaerially exposed as a result of glacioeustasy. (**c**) Beads-on-a-string pattern from Tinian Island, Marianas, the caves are exposed as a result of tectonic uplift; in both (**b**) and (**c**), the caves show the horizontal control of a past freshwater lens position.

over 3,000 m^2 of aerial footprint, developed during the last interglacial sea-level highstand (MIS 5e, 124–115 ka) (Labourdette et al., 2007). Because the caves form in the discharging lens margin, they tend to have a limited vertical extent commensurate with a thin lens; however, the caves have a broad aerial footprint that extends laterally along the coast, but not deep into the interior, as cave genesis is restricted to the lens margin area (Figure 2c). The flow within the caves is laminar, and the chambers are irregular and maze-like (Figure 2), a pattern known as ramiform (Palmer, 2007). The caves form without humanly passable entrances, and they become accessible only after surface erosion or collapse has exposed them. Once exposed, the caves form a pattern known as "beads on a string" (Mylroie, 2013), successive chambers that occur along a horizontal datum coincident with sea-level position at the time of cave genesis (Figure 3). Undiscovered caves present a collapse risk, but as their position is predictable based on current or past freshwater lens margin position, that risk can be mitigated (Lace and Mylroie, 2013).

Estuaries and coastal karst

Estuaries create two important environments that have a direct impact on coastal karren and karst features such as flank margin caves (Mylroie and Mylroie, 2009). First, estuaries are protected from wave energy, compared to the open sea, so flank margin caves survive longer than on open coastlines. Second, an estuary acts as a physical incursion into the land mass, which creates a shortcut for groundwater discharge; water can flow "sideways" as



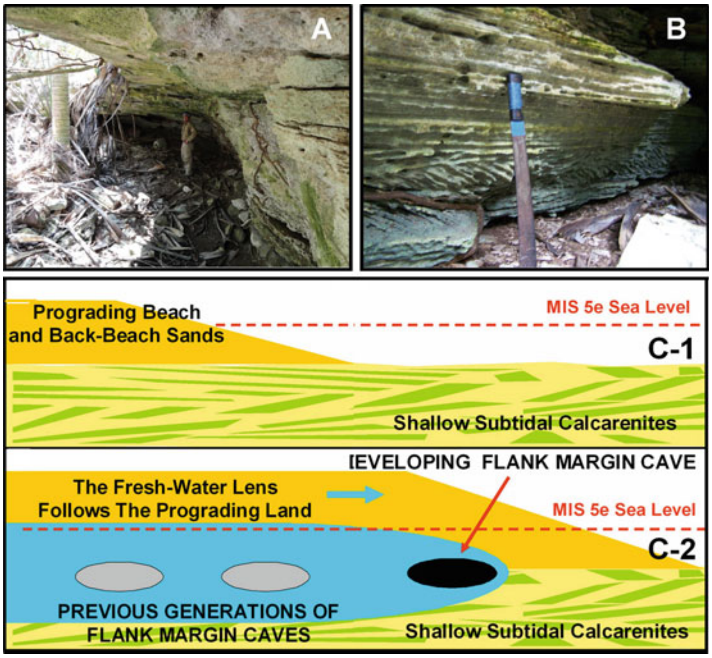
Karst Processes and Estuarine Coastlines, Figure 4 Sea caves and hybrid caves. (**a**) Coastal cliff and sea cave, northwest coast of Barbados, demonstrating wave action at work. (**b**) Animal Flower Cave, northwest coast of Barbados, 100 m left (north) of image (**a**). The cave originated as a flank margin cave but is now subject to littoral processes, overprinting it to a hybrid cave condition.

a shorter route to an adjacent embayment rather than continuing as diffuse flow to a distant coast. Mixing dissolution is focused along the periphery of the estuary, with a consequent appearance and growth of flank margin caves.

Where estuary coastlines are rocky and cliffed, sea or littoral cave development can occur in any lithology strong enough to support cliff development (Waterstrat et al., 2010). Sea caves form primarily by mechanical action of wave impact (Figure 4a), although bioerosion may contribute to a certain extent. In soluble rocks such as limestone, the development of flank margin caves just within the flank of the land creates a situation where wave attack can breach or open these dissolutional caves. Once breached, littoral processes can begin to erode the flank margin cave interior (Figure 4b), to produce a hybrid cave (Machel et al., 2011), one developed by overprinting a cave formed by one process (dissolution) with cave voids produced by a second process (wave erosion). Successful discrimination of eroded flank margin caves from simple sea caves is important. While both cave types can determine paleo-sea-level position, based on their appearance at a common elevation datum, the identification of a breached flank margin cave indicates a greater degree of overall coast retreat or denudation than a sea cave does (Waterstrat et al., 2010). Flank margin cave development is also dependent on the characteristics of the freshwater lens at the time of cave formation and can provide information regarding inland paleo-hydrological conditions that sea caves cannot (e.g., Mylroie et al., 2008a).

Gently sloping limestone coastlines can also contain actively forming flank margin caves. Carbonate sediments lithify quickly in the meteoric environment. As carbonate sediments are deposited, and subaerially exposed, the initial steps for lithification into limestone can begin. The presence of a freshwater lens margin allows cave development to similarly occur. The type of cave voids developing within the depositional environment, at the same time that carbonate deposition is occurring in adjacent areas, is called the syndepositional cave (Mylroie, 2013). During the last interglacial sea-level highstand (MIS 5e) in the Bahamas, longshore transport of carbonate sediment created prograding strand plains that produced land extensions into estuaries and lagoons. As this progradation occurred, the freshwater lens extended into the strand plain, and dissolution occurred at the lens margin. As the strand plain continued to prograde seaward, the freshwater lens margin continued to advance, and a succession of small dissolutional voids, or flank margin caves, was produced as syndepositional caves (Figure 5). Because the lens margin was stable at one position for only a brief amount of geologic time, the caves that formed were similarly small and simplistic, compared to flank margin caves developed on a rocky coast where the lens margin was stable for the entire sea-level highstand. A whole category of cave in the Bahamas, known as the banana hole (a local native term) – small, oval, and lenticular caves – has formed this way, restricted only to the last interglacial fossil strand plains (Mylroie et al., 2008b). Cave densities reach over 3,000 per km^2 (Harris et al., 1995). As the position of the top of the strand plain, relative to the freshwater lens contained within in it, is only a few meters, these banana holes have a thin roof which is susceptible to collapse, creating a surface sinkhole feature in which specialty crops are commonly grown (hence the name banana hole; Figure 5a). These features are a significant land-use hazard, as they can be present in great abundance and, if not yet exposed by collapse, present a significant risk. Their restriction to fossil Pleistocene strand plains allows areas of risk to be determined.

As a result of glacioeustasy, estuaries developed in limestones have seen sea levels that have ranged over 100 m during the Quaternary. Estuaries were repeatedly flooded, drained, and re-flooded as sea level migrated, a pattern well established in the Bahamas. In tectonically active areas, such as Barbados or Curaçao, the sea-level record is complicated by a tectonic uplift overprint. During sea-level lowstands, surficial processes dominated,



Karst Processes and Estuarine Coastlines, Figure 5 Banana holes as syndepositional caves, Bahamas. (**a**) Partially collapsed banana hole, note large tree to the left. (**b**) Late Pleistocene (MIS 5e) wall rock of a typical banana hole, showing the transition from subtidal herringbone facies upward into planar beach facies, typical of a prograding strand plain. (**c**) Diagrammatic model displaying how prograding strand plains allow the freshwater lens to similarly prograde (**c-1** to **c-2**), with the flank margin cave speleogenetic environment migrating as well; caves form during each pause in progradation but are abandoned when progradation resumes. Collapse of these thin-roofed voids results in the wide fields of banana holes known from the Bahamas.

and stream valleys were incised into coastal terrain. When sea level returned and flooded these valleys to create estuaries, mixing-zone dissolution resulted in flank margin cave development along the carbonate estuary periphery. In tectonic islands, uplift carried these estuaries above sea level. On Barbados, that uplift has lifted these paleo-estuaries above any possible glacioeustatic sea-level highstand position, and the valleys have been reoccupied by fluvial processes. These valleys are called gullies locally and have been previously interpreted to be collapsed stream caves because of open cave chambers and exposed speleothems (stalagmites, stalactites and flowstone, etc.) found on the gully walls (Machel, 1999). These caves have been reinterpreted as flank margin caves (Figure 6), now exposed by scarp and slope retreat (Kambesis and Machel, 2013). The caves fit the morphological pattern of flank margin caves and lack evidence of development by turbulent water flow (as stream caves would display), and the gullies containing these cave fragments are wider than any known stream caves on Barbados (e.g., Harrisons Cave or Coles Cave, each over 1 km long). In other words, if the gullies were collapsed stream caves, the scarp and slope retreat necessary to give the gullies their current width should have removed all evidence of the initial stream cave, including speleothems. Similar flank margin cave patterns can be found in the bokas of Curaçao (Kambesis et al., in press) and the *calas* of Mallorca (Mylroie et al., in press), rock-walled coastal reentrants that lead landward to in-feeding fluvial systems (Figure 6). The caletas of the Yucatan (Figure 7) represent a variation of karst reentrant produced by conduit-flow stream caves entering the coastal environment (Kambesis and Coke, 2013).

When an estuary has been uplifted beyond current and past sea-level positions, erosional processes can remove surficial marine deposits and weathering features (such as wave cut notches) that would otherwise allow these valleys to be identified as paleo-estuaries. The existence of flank margin caves on the periphery of these valleys indicates that they once held a marine water body, which helped drive mixing-zone dissolution. The same slope and scarp erosion that removed the surficial marine signature has opened the caves that developed under the flank of the land, allowing a more accurate interpretation of the valley's history as an estuary to be determined. In this manner, paleo-estuaries, especially those associated with the rapid sea-level fluctuations of the Quaternary, can be recognized when traditional evidence has been eroded away.

Soluble rock as an estuary coastal aquifer

The discharge of aquifers into estuaries has been called the subterranean estuary (Moore, 1999), where chemical reactions between marine water and aquifer material alter the composition of the discharging water. In addition to the mixing-zone dissolution previously discussed, the mixing

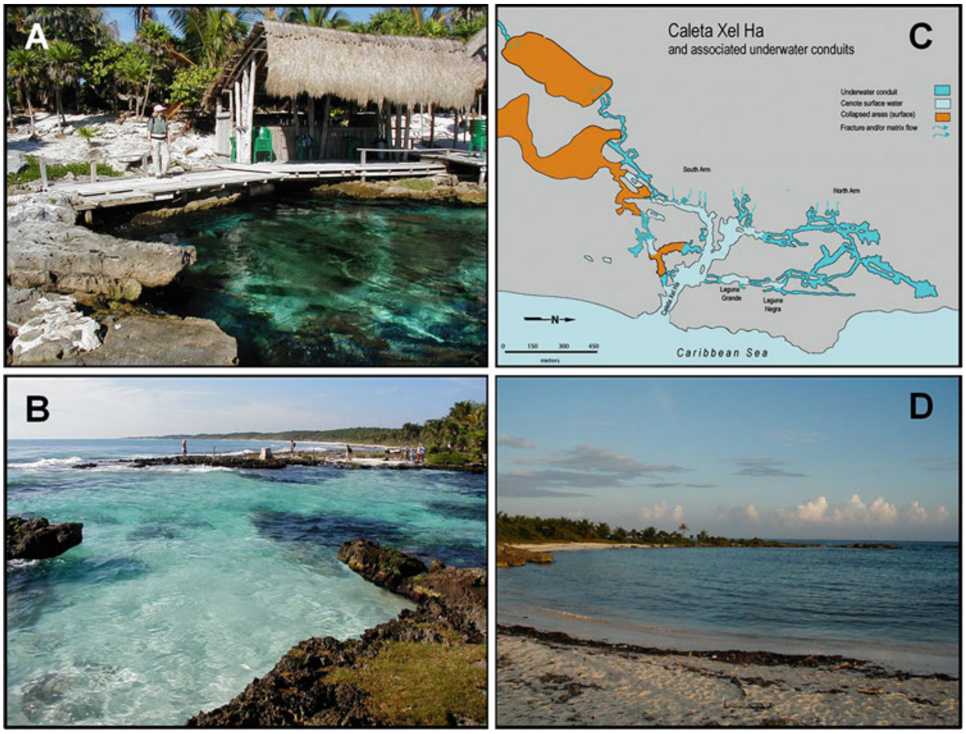
Karst Processes and Estuarine Coastlines, Figure 6 Coastal reentrants and karst processes. (**a**) A boka from Curaçao, showing wall collapse from failure of MIS 5e flank margin caves. (**b**) Flank margin cave preserved in the wall of a cala, Cala Figuera, Mallorca. (**c**) Map of Sailors Gully, Barbados, showing relict flank margin caves formed when glacioeustasy flooded a stream valley. (**d**) Flank margin cave remnant, Sailors Gully, Barbados.

zone is also a preferential site of diagenesis (Back et al., 1979), the lithological maturation of the rock (see below). The influx of nutrients, both natural and artificial, from coastal aquifers into estuaries has long been recognized (Moore, 1999).

In the low latitudes, where limestone deposition is ongoing, the carbonate rocks are necessarily young and have been subjected to minimal diagenetic modification. The earliest alteration is the inversion of primary depositional aragonite, a polymorph of $CaCO_3$, to its more stable form, calcite (also $CaCO_3$). This change in crystal structure, without a change in composition, releases trapped cations, notably Sr^{++}, which does not fit well into the calcite structure. Freshwater discharging from emergent very young limestones into an estuary will contribute a high Sr background compared to discharge at a later geologic time when the aragonite to calcite inversion process has gone to completion. Aragonite is also more soluble than calcite, and karst processes therefore work very fast in young limestones (Larson and Mylroie, 2013).

The young carbonate rocks in subaerial positions are called eogenetic, when buried and undergoing major diagenesis, they are called mesogenetic, and when returned by uplift to the earth's surface as diagenetically mature carbonate rocks, they are known as telogenetic (Choquette and Pray, 1970). Mesogenetic carbonate rocks, because of their deep burial position, are of no consequence to the discussion of estuaries. However, the difference between eogenetic and telogenetic carbonate rocks located in estuaries is significant. Eogenetic rocks retain most of their original depositional primary porosity, commonly ~30 %, and fluid flow is laminar diffuse flow across the volume of the rock. Telogenetic carbonate rocks retain minimal primary porosity and fluid flow is focused along joints, fractures, and bedding planes (Mylroie and Mylroie, 2013). Conduit flow is not restricted to telogenetic rocks. Where groundwater discharges are high, conduit flow can develop in eogenetic rocks (Mylroie and Mylroie, 2007). The advective circulation of seawater through conduits in eogenetic carbonate rocks has been documented along the Caribbean coast of the Yucatan Peninsula (Figure 7) and demonstrates a cavernous zone which extends almost 10 km inland and expresses a true subterranean estuary (Beddows et al., 2007).

Freshwater aquifer discharge into an estuary from carbonate rock will be very different depending on the diagenetic maturity of the rock. Eogenetic carbonate aquifers will commonly provide relatively uniform diffuse flow



Karst Processes and Estuarine Coastlines, Figure 7 Caletas as coastal reentrants, Quintana Roo State, Yucatan Peninsula. (**a**) Outflow of conduit freshwater into a coastal reentrant or caleta. (**b**) The outer area of the image in (**a**), displaying the rocky coast containing the caleta. (**c**) Map of a conduit system as it reaches the coast to create a caleta, initially as result of cave collapse. (**d**) A mature caleta, in which littoral action has created a crescent-shaped bay with beaches and rocky outcrops.

to the estuary margin (flank margin caves are mixing chambers embedded within this fluid flow system and have minimal impact on flow). Aquifer storage will be large. Particulate matter will be substantially filtered out, and slow flow velocities will give time for bioremediation of contaminants. In contrast, the focus of flow to the planar structures of telogenetic carbonate rocks (joints, fractures and bedding planes) leads to the development of turbulent flow pathways, which result in discharge to selected points along the estuary perimeters, rapid flow with little filtration, and minimal aquifer storage. Conduit systems in eogenetic rocks, as in the Yucatan, create an interesting duality, with large amounts of storage in the matrix, but with well-established conduits containing turbulent flow routes capable of rapid water transmission. The impact on coastal ecosystems should be very different, with contaminant transport more of an issue in conduit-flow systems. Anchialine cave systems and their biota are sensitive to the nature of the voids enclosing them in rocky coastal environments, and large variations in species diversity are expected (Mylroie and Mylroie, 2011; van Hengstrum et al., 2011).

In terms of determining estuary water budgets and salinity gradients, the contribution of karst groundwater flow, either as diffuse flow from eogenetic carbonates or as focused flow from telogenetic carbonates, can be easily missed. Karst watersheds are by definition cryptic, with substantial surface flow diverted underground, while in teleogenetic rocks the flow patterns do not differ much from surficial stream channels in terms of discharge rates and volumes. Proper characterization of coastal estuary waters in carbonate settings requires taking into account the unique processes associated with karst. The investigator also needs to recognize the diagenetic maturity of the carbonate rock.

Summary

Estuaries with part, or all, of their perimeter contained by soluble rocks will display differences from those estuaries developed in non-carbonate rock or sediments. Flank margin caves are the most notable difference and can create coastline irregularities and specialty habitats. These caves can also provide evidence of paleo-estuary conditions when other evidence has been removed by erosion. Freshwater input to the estuary may approach conditions similar to typical diffuse groundwater flow if the soluble rocks are eogenetic. If the soluble rocks are telogenetic, then conduit-flow additions to the estuary can occur at multiple discrete points along the estuary perimeter that may not be obvious to casual observation. Accurate assessment of an estuary developed in carbonate rocks cannot be done without an awareness of how karst processes are participating in coastline development, habitat placement, and freshwater flow.

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Cross-references

Climate Change
Coastal Lagoons
Coastal Landforms
Diagenesis
Estuarine Geomorphology
Halocline
Shoreline Changes
Submerged Coasts
Water Resources and Climate Change

KELP FORESTS

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Synonyms

Giant kelps; Giant seaweed; Kelpos; Sargazo Gigante

Definition

Kelp is the common name given to the large algae belonging to the class Phaeophyceae (brown algae) in the order Laminariales or the morphologically similar fucoids. Both canopy types include fronds suspended in the water column by some form of flotation (*Macrocystis pyrifera*, *Egregia menziesii*, *Cystoseira osmundacea*, *Sargassum* spp), fronds supported above the substratum by semirigid stipes (*Pterygophora californica*, *Eisenia arborea*), and fronds lying on or immediately above the substratum (*Laminaria farlowii*, *Agarum fimbriatum*) (McPeak et al., 1988; Lobban and Harrison, 1994; Graham et al., 2008).

Structures

The giant kelp (*M. pyrifera*) consists of one thallus (or body) formed by the holdfast that holds it fast to the substrate. The holdfast also provides a home for many marine creatures. It presets many stipes which are elongated stemlike structures. The blades are formed along the stipe. Pneumatocysts occur at the connection of the stipe and the blade; these are gas-filled bladders that provide buoyancy to lift the fronds toward the surface. Other kelps, such as *Nereocystis luetkeana*, have only one big blade, and still others, such as *Eisenia arborea*, lack a bladder. The part of the organism that lies on the surface of the sea is called the canopy.

Reproduction

The life cycle of the large kelps involves an alternation of generations between the adult sporophyte (diploid) that produces spores and the microscopic gametophyte (haploid) that produces sperm or eggs. Sexual reproduction results in the beginning of the diploid sporophyte stage, which develops into a mature individual. In Fucales, meiosis takes place in the conceptacles and gamete fertilization on the conceptacle surface. The propagules that disperse are diploids.

Environmental factors

Kelps are affected by many biotic (grazers and competition) and abiotic (nutrients, substrate, light, and temperature) factors. Large kelps require nutrient-rich water with temperatures between 6 °C and 14 °C. They are known for their high growth rate. The genera *Macrocystis* and *Nereocystis* can grow as fast as 50 cm per day, ultimately reaching 30–80 m in size.

Conclusions

Kelp forests exhibit very complex ecological relationships among their associated species in which complex trophic networks develop based on species abundance. Kelp forests are vital areas where many fish species recruit, using them as nursery and reproductive habitat. Giant kelps can be easily harvested because of their surface canopy. Initially they were harvested to produce soda ash but now are used to produce alginate, a carbohydrate that thickens products or forms gels (ice cream, jelly, salad dressing, or pet food). Alginate is also used for dental impressions and beer production as well as many other products. In other uses, kelps are good sources of biodiesel, but the lack of extensive culture techniques prevents their wide use for this commercial purpose.

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Cross-references

Macroalgae

L

LIGHT DETECTION AND RANGING (LIDAR)

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Synonyms

Airborne laser scanning; Airborne laser swath mapping; Airborne laser terrain mapping; Laser altimetry; Laser radar; Laser ranging

Definition

Light detection and ranging (lidar) is an active sensing technology that pulses a laser and records the energy reflected from a distant object to measure its distance. The basic principle of operation is analogous to radar except it uses laser light energy instead of radio waves.

Description

Lidar systems consist of a laser transmitter and a receiver to collect the reflected energy. By precisely measuring the round-trip time of travel of the laser pulse as it propagates from the laser transmitter to the object and back, the range from the lidar sensor to the object can be accurately determined (Figure 1). For surveying and mapping purposes, lidar systems typically include a scanning mechanism, such as an oscillating mirror, to direct the laser pulse and image a scene (Shan and Toth, 2009). Lidar mapping systems are often mounted in mobile platforms, such as an airplane (commonly referred to as airborne lidar), to scan large areas. Range measurements for each laser pulse are integrated with measurements of the platform's position and orientation along with the scanning angle at the time of the pulse to derive three-dimensional, geo-referenced coordinates (positions) of points on the surface. The end product of a lidar mapping survey is a point cloud of irregularly spaced *x*,*y*,*z* values providing a 3D representation of the scanned

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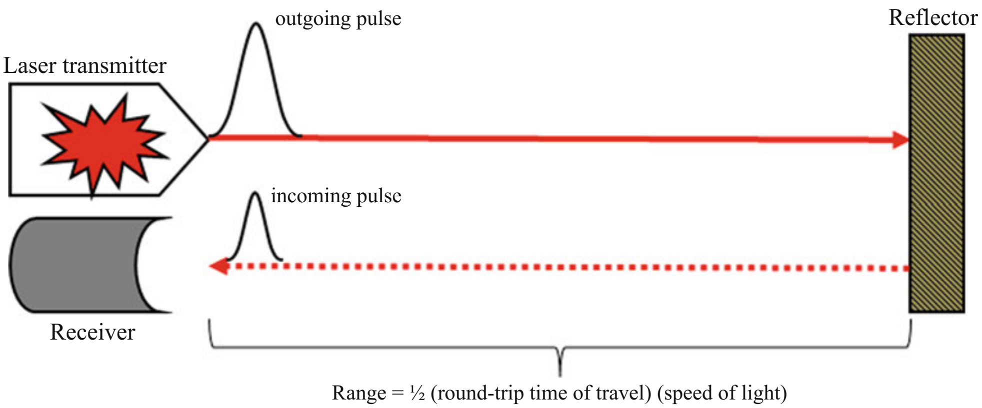

Light Detection and Ranging (LIDAR), Figure 1 Basic principle of pulsed lidar. The lidar system measures the round-trip time of travel of the laser pulse as it propagates from the laser transmitter to the reflective object and back.

region (e.g., the ground and land cover). These data are often used to derive digital elevation models (DEM) or digital surface models (DSM) of the imaged terrain.

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Cross-references

Airborne Laser Terrain Mapping (ALTM)

LITTORAL CORDON

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Synonyms

Sandbar

Definition

Littoral *cordon* (cord, belt) is a Spanish term from the genetic classification of simple coastal forms (Bores, 1978) which together with littoral *flecha* (arrow) is a term used to describe the shaping of most barrier islands (Finkl, 2004).

Genesis

Barrier islands can be formed in a number of ways (King, 1972; Williams, 1982), with wave and sediment processes playing significant roles. The linear momentum of the transversal (orthogonal) wave generates a bar in the breaker zone that results in a radical modification of hydrodynamic conditions. The bar may become a littoral cordon if the alongshore (parallel) linear momentum is relatively minor. If, in contrast, this longitudinal component is significant, the bar tends to advance in the direction of littoral transport giving rise to a spit. Both cordons and spits may become barrier islands.

Climatic conditions can cause short-term increases in sea level via (1) low-pressure suction (storm surge), (2) wind setup, and (3) wind-wave setup. The last two processes are always interrelated and increase together, though with a delay. Further, both are coupled to the first process, though not jointly. Maxima of the first or third processes are not concurrent with maxima of the second, though they are additive in a maximum storm, which is precisely when storm bars are created. Storm bars are deeper than bars formed during more quiescent conditions. Storm bars may also attain higher elevations. At times, when storm waters recede, storm bars may remain above sea level.

A bar with insufficient thickness, elevation, or sedimentary consistency is often temporary. This can happen when sea level is not extraordinarily high. However, if sea level is high enough and the nature of the sedimentary system supports it, a sandbar can survive the annual cycle and become a permanent formation once it is colonized by biota.

There are broad morphological and genetic analogies between littoral cordons and spits, such as the wave action under which the features evolve. There are also radical differences between the features. One pronounced difference is the relative importance of parallel littoral transport.

Littoral cordons and spits facilitate the subsequent infilling of an intracoastal channel or bay. This leads to the eventual extension of the littoral plain and the creation of residual littoral lagoons that have some degree of permanence. Up to three successive barrier islands that have been added to the mainland on the Guadiana River delta front (southwestern Spain) may be attributed to the genesis of littoral cordons. This process may also have been active in the past at the delta front that forms the outlet of the Rhine, Meuse, and Elm rivers in the Netherlands.

Relict littoral cordons may remain submerged under rising sea-level conditions, and they may shape sedimentary ridges in front of many coastal areas. These features are obviously very different from reefs or actual tectonic ridges.

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Cross-references

Bar
Barrier Island
Coastal Bays
Spit

LITTORAL ZONE

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Synonyms

Intertidal zone

Definition

In an estuary, the littoral (intertidal) zone is defined as the area between the high and low tide marks (Kennish, 2001). In an ocean, many scientists define the littoral zone more broadly as the area extending from the shore to the edge of the continental shelf and encompassing the supralittoral, eulittoral, and sublittoral zones. As such, most of the littoral zone lies in the shallow water region at sea where light penetrates to the bottom.

Characteristics

The intertidal zone of estuarine and marine environments is highly variable in physical and chemical conditions. Organisms inhabiting this zone must be able to withstand multiple stresses associated with periodic exposure, strong wave action and turbulence, shifting sediments, and susceptibility to predation from both terrestrial and aquatic organisms (Kennish, 2001; Day et al., 2012; Levinton, 2013). Many organisms found along high-energy, rocky intertidal zones are sedentary and firmly attached to a substrate.

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Cross-references

Shoreline
Sublittoral Zone

LOBSTER MIGRATION

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Definition

Lobsters frequent estuaries and bays throughout their range and exhibit seasonal movements or migrations.

Background

Estuarine habitats are demanding marine environments for crustaceans such as lobsters because these areas are frequently subject to dramatic fluctuations in salinity and temperature (Watson et al., 1999). Lobsters that reside in estuarine systems therefore optimize their survival in this complex habitat through both physiological (changes in osmoregulation) and behavioral adaptations (behavioral thermoregulation) (Crossin et al., 1998; Charmantier et al., 2001). While adult clawed lobsters (*Homarus americanus*) prefer higher salinities (20–25 psu) over lower ones (10–15 psu) (Jury et al., 1994), females appear much more sensitive to reduced salinity. Thus, there are often differential movements by males and females in estuaries in response to salinity or temperature gradients, often setting up skewed sex ratios (Howell et al., 1999; Jury and Watson, 2013).

Lobster migrations are generally considered direct locomotory movements over a discrete time period and toward environmental conditions that are different (see Herrnkind, 1980). For example, Caribbean spiny lobsters (*Panulirus argus*) typically exhibit fall movements from bays and shallow coastal areas into deeper waters (Herrnkind, 1980). A detailed examination of the seasonal movements of clawed lobsters (*H. americanus*) into a New England estuary (Great Bay, NH) showed that lobsters moved into the estuary in the spring when salinities were >15 psu and had a tendency to move downriver (toward the coast) for the remainder of the year (Watson et al., 1999). Because lobsters avoid areas of suboptimal temperature and salinity, they may move and concentrate in areas with more favorable environmental conditions.

The role of estuaries for migrating lobsters includes the use of elevated summertime water temperatures for growth, maturation, or reproduction thereby inducing faster molting and as a possible refuge (Moriyasu et al., 1999).

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Cross-references

Thermal Biology
Water Quality

M

MACROALGAE

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Synonyms

Seaweeds

Definition

Macroalgae, commonly termed seaweeds, are multicellular, macroscopic benthic algae belonging to green (phylum Chlorophyta), red (phylum Rhodophyta), and brown (phylum Phaeophyta) taxonomic groups. They are found in estuaries throughout the world attached to substrates or drifting in the water column. Benthic macroalgae generally attain highest abundance in shallow estuaries and coastal lagoons where the estuarine floor lies within the photic zone.

Characteristics

Macroalgae exhibit a range of body types, including filamentous, leafy, sheetlike, tubular, branching, and feathery forms. They can reproduce sexually or asexually. In shallow estuaries and coastal lagoons, macroalgae often attain high diversity; for example, more than 100 species of macroalgae have been recorded in Barnegat Bay-Little Egg Harbor, a coastal lagoon in New Jersey (USA) (Kennish, 2001). In addition, they often generate high biomass, particularly during bloom events, when they typically form thick mats covering extensive areas of the estuarine floor or wetland surface (McGlathery et al., 2007). In nutrient enriched waters, macroalgal populations may reach a peak biomass of more than 0.5 kg m^{-2}, with canopy heights exceeding 0.5 m (McGlathery, 2001). Temperature, light, nutrients, and grazing pressure are mechanisms that control the rates of net production in macroalgae (Valiela et al., 1997), which are important habitat formers for many estuarine organisms. While some macroalgae inhabiting estuaries belong to a drift community unattached to any substrate, others attach to rocks, bulkheads, piers, and other hard substrates by root-like holdfasts or basal disks. On mudflats, epipelic, epipsammic, or drift populations typically predominate.

In estuaries, macroalgae disperse as free-floating forms or as microscopic propagules (spores, gamete, and zygotes), and hence currents play a significant role in their distribution. Macroalgae frequently display a distinct zonation in intertidal zones, notably on rocky shores. This distribution is largely controlled by tidal variation, competition for light, desiccation (emersion), and grazing.

Common macroalgal taxa in estuaries include the following: *Cladophora*, *Enteromorpha*, and *Ulva* (Chlorophyta); *Ascophyllum*, *Fucus*, and *Sphacelaria* (Phaeophyta); and *Ceramium*, *Gracilaria*, and *Polysiphonia* (Rhodophyta).

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M.J. Kennish (ed.), *Encyclopedia of Estuaries*, DOI 10.1007/978-94-017-8801-4,

Cross-references

Algal Blooms
Macroalgal Blooms
Toxic Blooms

MACROALGAL BLOOMS

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Synonyms

Seaweed blooms

Definition

Macroalgal blooms are rapid accumulations of macroalgal populations that form dense canopies in benthic communities of estuaries and coastal lagoons (Valiela et al., 1997; Anderson et al., 2010). They are manifested by a marked increase in macroalgal biomass (often >0.5 kg m^{-2}; McGlathery, 2001) that falls outside normal seasonal cycles of abundance. The percent cover of intertidal/subtidal benthic habitat is also used as a measure of macroalgal bloom occurrence. Canopy heights may exceed 0.5 m (McGlathery, 2001).

Characteristics

The proliferation of bloom-forming "ephemeral" macroalgae appears to be on the rise in shallow estuarine and coastal lagoonal environments due to the increase in nutrient enrichment and eutrophication of these water bodies observed for decades in temperate and tropical regions (McGlathery, 2001; Kennish and Paerl, 2010). While macroalgal blooms lack the direct chemical toxicity of toxic phytoplankton blooms, they have a broader range of ecological effects and typically last longer (Valiela et al., 1997). For example, Gordon and McComb (1989) reported that a bloom of *Cladophora* lasted for about a dozen years in Peel Harvey Estuary, Western Australia, and Valiela et al. (1992) documented blooms of *Cladophora* and *Gracilaria* exceeding 20 years in Waquoit Bay, Massachusetts (USA). Macroalgal "nuisance" blooms are detrimental to seagrass beds because they block or attenuate light (Burkholder et al., 2007; Kennish et al., 2010). Persistent macroalgal blooms therefore often result in the displacement and loss of essential seagrass habitat by macroalgal populations (McGlathery, 2001; Kennish et al., 2010). These blooms may cause other impacts such as the increase in sediment organic matter and oxygen depletion (hypoxia/anoxia), decrease in redox potential, and increase in sediment sulfide concentrations as the macroalgal layer decays. Hydrogen sulfide concentrations also typically increase and can create a hazardous environment to humans in proximity to the bloom sites.

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Cross-references

Algal Blooms
Macroalgae
Toxic Blooms

MACROFAUNA

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Definition

Macrofauna are estuarine and marine organisms visible to the naked eye ($>$ 0.5 mm) that commonly inhabit the benthos, where they can be found buried in sediment or attached to a fixed substrate (rocks, reefs, rhodolith, etc.). Some macrofauna may also be observed moving on the sediment surface as epifauna or in the water column. Macrofauna that are often abundant in the benthos include flatworms (Platyhelminthes), sipunculids, nematodes, polychaetes, isopods, amphipods, stomatopods, pycnogonids, chitons, bivalves, gastropods, echinoderms, bryozoans, and urochordates. Macrofauna that are commonly found in the water column include jellyfish, ctenophores, salps, and some larvae of benthic invertebrates such as sponges, mollusks, polychaetes, and crustaceans that exceed 0.5 mm in length. Benthic macrofauna can play an important role in the detoxification of estuarine and marine environments because they store and transform many types of pollutants (Greenway, 1995; Schlacher and Wooldridge, 1996; Beaumont et al., 2007).

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Cross-references

Benthic Ecology
Infauna
Meiofauna
Microfauna
Soft Sediment Communities

MACRONUTRIENTS

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Synonyms

Inorganic nutrients; Limited nutrients

Definition

Macronutrients are chemical elements required by an organism in relatively large amounts (e.g., N and P required by plants). They are seldom in excess of a few micromolar concentrations.

Description

Inputs. Most biologically important compounds entering an estuary are derived from riverine sources. After the industrial era, the use of fertilizers and fossil fuels increased dramatically, leading to a substantial increase in nutrient concentrations in estuarine systems.

Demand. Estuarine primary production requires the availability of a variety of nutrient elements. The chemical elements essential for primary producers are carbon (C), oxygen (O), nitrogen (N), phosphorus (P), and silicon (Si). Carbon, nitrogen, and phosphorus are the most important elements for forming molecules vital to organisms (Day et al., 2012).

Stoichiometry. Nutrient elements accumulate in phytoplankton to a relatively constant internal composition. As described by the Redfield ratio, the marine phytoplankton intracellular O:C:N:P ratio is 138:106:16:1. Inadequate levels of a nutrient element in the environment can limit an organism's metabolic activities.

Limitation. Carbon is abundant in the marine environment (carbonate, bicarbonate, and carbon dioxide) and thus is rarely limiting. The N:P ratio best represents the macronutrient limitations (Redfield et al., 1963). According to Liebig's Law of the Minimum, the concentration of a nutrient present at the lowest relative levels required for an organism's growth is typically the limiting factor, which explains why the N:P ratio is a limiting factor compared with carbon (Bianchi, 2007).

Estuarine Behavior. Certain estuarine microorganisms may have variable internal N:P ratios because the internal composition reflects the ratio in the environment and N and P concentrations are highly variable in estuaries. Therefore, changes in bioavailability of such elements can significantly limit organic carbon production. Nitrogen is the limiting nutrient in most estuaries because it is assimilated more rapidly and in greater amounts than other nutrients. It also has a more complex biogeochemical cycle than phosphorus. Reactive dissolved silicon is a key component in the composition of diatom and silicoflagellate shells as well as exoskeletons of other organisms, such as certain sponges. Reactive dissolved silicon is assimilated by diatoms at ratios similar to those at which N is assimilated.

Eutrophication. Eutrophication caused by nutrient enrichment (i.e., N and P) is a major problem in many estuaries (Kennish, 1997). Excessive loading of N to estuarine waters can result in P limitation in systems that are generally considered to be N limited.

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Cross-references

Algal Blooms
Bioavailability
Ecological Stoichiometry
Nitrogen
Nutrient Limitation
Phosphorus

MANAGED REALIGNMENT

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Synonyms

De-embankment; Managed retreat; Setback

Definition

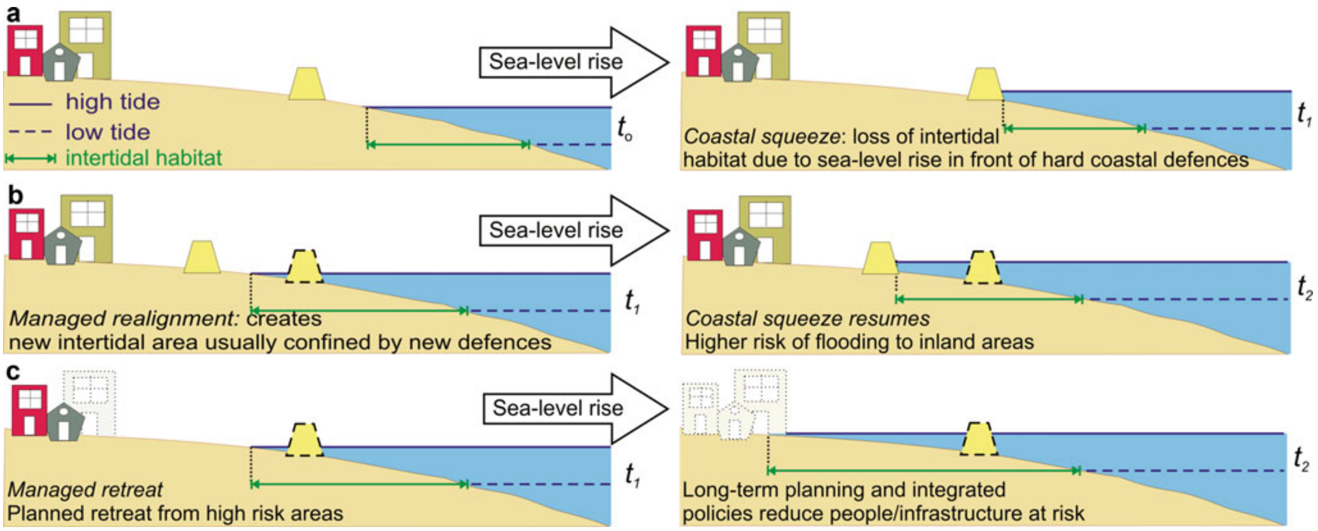
Managed realignment most often involves the planned breaching or removal of coastal defenses to create new intertidal habitats aiming to improve flood risk management with added environmental value. Managed realignment is usually implemented in low-lying estuarine or open coast sites, and may require the construction of a new line of defenses to control flood risk. Hence, the expression "managed realignment" may refer to the relocation of both the coastline and the flood defense line. An overview of the different definitions used in the literature is provided in Esteves (2014).

A shift from the traditional 'hold-the-line' approach of coastal protection

Managed realignment is one of the soft engineering approaches to coastal protection. By working with coastal processes, managed realignment aims to increase the sustainability of coastal protection while at the same time reducing adverse environmental impacts normally associated with hard engineering (French, 1997). For centuries, hard engineering structures have been built to protect assets at the coast from erosion and flooding events. These hard structures have created a legacy of coastal management problems, which are now considered unacceptable, including the loss of intertidal habitats due to coastal squeeze (Figure 1a).

The two most important climate change effects predicted for coastal areas are sea-level rise and more frequent and intense extreme weather events (e.g., IPCC, 2007). Climate change impacts are likely to increase the risk of flooding and erosion posing a greater threat to people and infrastructure at many coastal locations. It is therefore required that coastal defenses are upgraded and more frequently maintained so they continue to provide the current level of protection to inland areas in the future. The consequent increase in costs of coastal protection has made the traditional "hold-the-line" approach unsustainable in many coastal areas. Managed realignment is an increasingly popular alternative to address both the economic viability and the environmental sustainability of coastal protection, especially in reclaimed estuarine areas (French, 2001).

Unlike coastlines "fixed" by hard coastal engineering, natural coasts dynamically respond to changes in accommodation space due to sea-level fluctuations or alterations



Managed Realignment, Figure 1 Schematic diagram representing: (**a**) coastal squeeze, the loss of intertidal areas due to rising sea levels in front of fixed coastlines; (**b**) managed realignment, the creation of new intertidal area and the return of coastal squeeze at sites where saltmarshes fail to develop; and (**c**) managed retreat, which integrates land-use planning and long-term risk reduction by creating new intertidal habitats and removing people and property from risk areas. Different moments in time are indicated by t_{0-2}.

in sediment budget. Saltmarshes, for example, depending on a number of interacting biotic and abiotic variables (e.g., the accommodation space and sediment supply), can migrate inland and accrete vertically, naturally adjusting to rising sea levels. These intertidal habitats provide a number of ecosystem services (e.g., Luisetti et al., 2011), such as natural coastal protection by dissipating wave energy (Möller et al., 2007), therefore contributing to reduced flood risk to inland areas and the associated cost of maintaining existing flood defenses.

Geographic distribution

The first managed realignment projects were implemented in France in 1981 and in Germany and the Netherlands in 1989 (Esteves, 2014). Managed realignment has since become increasingly popular in northern Europe (Mazik et al., 2010), especially in England (where the highest number of projects has been implemented), Germany, the Netherlands, Belgium, and France. A list of projects implemented in Europe, including their main characteristics, is available from the *ABPmer Online Managed Realignment Guide* (http://www.abpmer.net/omreg/). The main objectives and the way projects are implemented vary considerably between these countries.

In England, managed realignment is implemented to create intertidal habitat and to deliver more sustainable flood risk management, e.g., by reducing costs and aggregating environmental and amenity values (Esteves, 2013). In Germany, managed realignment sites are found along the coast of Lower Saxony (by the North Sea) and Mecklenburg–Western Pomerania (by the Baltic Sea), but the objectives differ across these two areas (Rupp-Armstrong and Nicholls, 2007). In Lower Saxony, managed realignment is usually implemented for compensation reasons (i.e., loss of intertidal habitats due to coastal development, port construction etc.). In Western Pomerania managed realignment often combines the need for improvement of flood defenses and creation of new intertidal habitats. In Belgium most projects have been implemented along the Scheldt Estuary through the mechanism of controlled reduced tide (Beauchard et al., 2011; Teuchies et al., 2012) for compensation of damage or loss of intertidal habitats.

Outside Europe, managed realignment projects exist but are not known as such, being difficult to ascertain how many already exist. Although the terms managed realignment and managed retreat are often used interchangeably in the UK (e.g., French, 2001), elsewhere managed retreat refers to the relocation of people and assets at risk (e.g., National Oceanic and Atmospheric Administration (NOAA) Office of Ocean and Coastal Resource Management; http://coastalmanagement.noaa.gov/initiatives/shoreline_ppr_retreat.html).

How does it work?

By allowing tidal waters to flow farther inland through breached defenses, managed realignment creates new intertidal areas (Figure 1b) and accommodation space for sediment deposition. It is expected that the realignment site will act as a sink for sediments, favoring the development of saltmarshes. The resulting wider intertidal profile provides natural coastal protection through the dissipation of wave energy (French, 1997), which tends to be significantly greater over saltmarshes than over un-vegetated intertidal flats (Möller et al., 2007). Saltmarsh development enhances local biodiversity and the sustainability of coastal protection and, therefore, is crucial for the success of managed realignment as a sustainable coastal management approach.

Information on the performance of managed realignment projects is still scarce as most projects do not benefit from systematic long-term monitoring (Spencer and Harvey, 2012). Although many gray literature reports have been produced by consultants contracted to conduct the design, implementation, and monitoring of the schemes, only few independent monitoring studies have been published in peer-reviewed journals. The existing articles indicate diverse findings on the development of saltmarshes at managed realignment sites.

Vegetation colonization at managed realignment sites is reported to occur rapidly, most commonly dominated by pioneer saltmarsh species, as reported in sites along the Blackwater Estuary in England. Garbutt et al. (2006) suggested that the low elevation of the Tollesbury site contributed to the dominance of pioneer saltmarsh recorded 6 years after the breaching of defenses. At Orplands Farm, 8 years after managed realignment, the site showed low species saturation index and was dominated by pioneer and low marsh species due to poor drainage and seed availability (Spencer et al., 2008). At Freiston Shore (The Wash, England), high sediment input favored rapid colonization by pioneer saltmarsh vegetation (Friess et al., 2012). However, sediment had originated from the erosion of adjacent established habitats caused by the unexpected growth of the tidal creeks at the breaches (Rotman et al., 2008).

In their analysis of saltmarsh re-creation in Europe, Wolters et al. (2005) observed that only 50 % of the expected species were found at sites smaller than 30 ha. The authors concluded that biodiversity increased at sites larger than 100 ha, where the largest range of elevations between mean high water of neap and spring tides occur. Many managed realignment sites in England and elsewhere are small (<20 ha), low-lying, and confined by a new line of coastal defenses. These characteristics compromise the sustainability of managed realignment sites, as the lifetime of the newly created intertidal habitats depends on whether sediment availability (and other variables) will allow vertical accretion at rates that will cope with rising sea levels (Esteves, 2013). If saltmarshes are not able to fully develop (e.g., due to the small size or low elevation of managed realignment (MR) sites), it is just a matter of time until water levels reach the new line of defenses and the new intertidal habitats are again lost due to coastal squeeze (Figure 1b).

Many managed realignment projects have re-creation of intertidal habitats as a primary objective. This approach is partially driven by the need to address statutory duties (e.g., the EU Habitats Directive) to take all necessary measures to avoid detrimental impact to designated conservation areas and provide compensation for loss of these habitats. However, recent studies have indicated that marshes created by managed realignment are "significantly impaired" in their ability to deliver ecosystem services when compared with natural systems (Spencer and Harvey, 2012) and do not meet the requirements of the EU Habitats Directive (Mossman et al., 2012). Ecosystem services valuation (Luisetti et al., 2011) concluded that managed realignment can be economically efficient at time frames longer than 25 years. However, results are site-specific and should not be generalized, especially when "complex social decisions" are involved (Luisetti et al., 2011), such as in areas where people and assets are at risk.

Managed realignment versus managed retreat

The focus of managed realignment projects oscillates between improved flood risk management and environmental objectives, often with a bias toward habitat creation. Usually, medium- to long-term effects on flood risk to inland areas are not clearly assessed, probably due to uncertainties on the type of intertidal habitat that will develop and how they will evolve through time. Where saltmarshes fail to develop, coastal squeeze resumes as sea level rises, posing a higher risk of flooding to people and property. Conceptually, managed realignment has great potential to (1) provide space for the creation of intertidal habitats, (2) provide natural defense against storms and rising sea levels, and (3) contribute to the achievement of EU directives (i.e., floods, habitats, and water framework). Esteves (2013) states that for this potential to be realized, it is necessary that managed realignment implementation (1) follows a long-term strategic plan that effectively integrates its multiple objectives (e.g., habitat creation, flood protection, and amenity), (2) has clearly defined local and national targets at known time frames, (3) benefits from systematic monitoring so performance can be adequately measured against targets, and (4) is evaluated based on evidence so adjustments to the strategy can be put in place where necessary.

In contrast with managed realignment, the main objective of managed retreat is the relocation of people and assets at risk. Implementation of managed retreat might include relocation of single structures at risk (e.g., the historic Cape Hatteras Lighthouse, USA) or a series of measures to reduce the number of people and property at risk (e.g., the compulsory purchasing of property at high risk adopted in France after the aftermath of the Xynthia storm of 2010). Implementation of such schemes is complex due to the range of public perception conflicts (e.g., Roca and Villares, 2012), in addition to institutional capacity and economic constraints. Managed retreat usually requires strong integration between long-term planning and the sustainability of risk reduction measures, which is often deficient in public administrations. However, challenging times require drastic changes and the only safe climate-proof response at all temporal and spatial scales is to reduce the number of people and assets at risk. As it is an effective mechanism to reduce risk from both climatic variability and extreme events, managed retreat has increasingly been implemented (or planned) in many locations worldwide.

It is important to note that, so far, managed realignment has been implemented only in rural areas. However, as flood defenses are moved further inland, a long-term strategy is required to prevent risk to inland areas becoming unacceptable. Managed retreat deals with development in hazard-prone areas and, combined with long-term planning, may be applicable to a range of urban and industrial areas. A more effective strategy to reduce the risk of flooding to people and property would involve long-term planning objectives with both managed realignment and managed retreat implemented in predefined time frames.

Summary

Managed realignment is a soft engineering approach that aims to create intertidal habitat (especially saltmarshes) through the artificial breaching or removal of flood defenses. The creation of intertidal habitats has two main aims: (1) to offset the loss of designated intertidal habitat (due to coastal squeeze and developmental pressures) and (2) to dissipate wave energy to offer sustainable coastal protection. Managed realignment is becoming a popular coastal management approach in northern Europe. As managed realignment is a relatively new approach, there is a need to better understand the short- to long-term effects on (1) local sedimentary processes, (2) inland flood risk and development of intertidal habitats (and associated biota), (3) and wider socioeconomic and environmental implications.

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Cross-references

Climate Change
Coastal Risks: Floods
Coastal Squeeze
Coastal Wetlands
Habitat Loss
Saltmarshes

MANGROVES

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Synonyms

Mangal

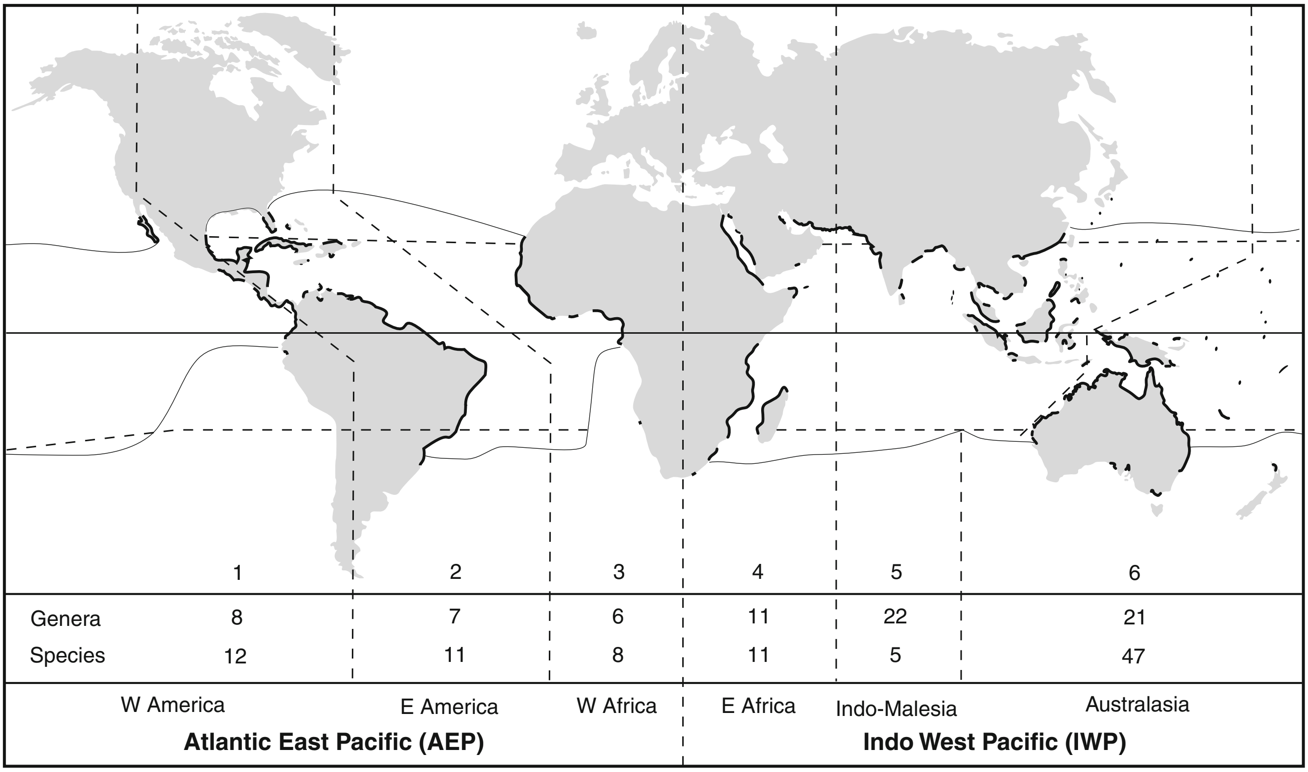
Definition

Mangroves are dicotyledonous woody trees and scrubs that grow above mean sea level to form intertidal forests along subtropical and tropical coasts.

Introduction

Mangroves are forested wetlands living along coasts within low latitudes. These tidal trees and scrubs occur on rocky and sandy shores, but they attain peak size and luxuriance in sheltered muddy areas where quiescent conditions foster establishment and growth of propagules (Lugo and Snedaker, 1974; Chapman, 1976). Forest establishment involves positive feedback in that the saplings and trees trap silt and clay particles brought in by tides and rivers, helping to consolidate the deposits on which they grow. This feedback process continues for the life of the forest until, eventually, the forest floor lies above the reach of tides. Over years and decades, terrestrial plants eventually outcompete and replace the mangroves assuming that geological and ecological processes are in equilibrium. The intertidal zone and its biota are highly dynamic and ever changing, disturbed often enough by weather events, such as storms and cyclones, disease, pests, and anthropogenic intrusions that the progression to terrestrial forest occurs infrequently along most coastlines. Mangroves occupy a harsh environment, subjected daily to tidal and seasonal variations in temperature, salinity, and anoxic soils, and are fairly robust and highly adaptable or tolerant to such changes.

Mangroves occur in a variety of coastal settings dominated by rivers (and the lack thereof), tides, and waves and develop and persist over timescales in which morphological evolution of coastlines occurs; they are pioneers colonizing newly formed mudflats, but they can also shift their intertidal position in the face of environmental change. In relation to changes in sea level, mangrove development can follow one of six patterns: (1) the mangrove surface accretes asymptotically until sediment accumulation raises the forest floor above tidal range – this pattern occurs when sea level is in equilibrium; (2) accretion of sediment keeps pace with a constant rise in sea level; (3) the forest floor accretes at times above tidal range when sea-level rise is irregular; (4) with episodic subsidence but with a stable sea level, the forest floor accretes back to the tidal range; (5) mangrove accretion continues at an irregular pace under conditions of episodic



	1	2	3	4	5	6
Genera	8	7	6	11	22	21
Species	12	11	8	11	5	47
	W America	E America	W Africa	E Africa	Indo-Malesia	Australasia
	Atlantic East Pacific (AEP)			Indo West Pacific (IWP)		

Mangroves, Figure 1 Global distribution of the world's mangrove forests and their biogeographic provinces. The *bolded lines* indicate mangrove coasts. The number of genera and species within each province is noted below the map (Modified from Alongi 2009).

subsidence but rising sea level; and (6) the forest floor is set back when there is no change in sediment volume with a rise in sea level (Woodroffe, 2003). Thus, mangroves are not static ecosystems, but ever changing, like the interface they occupy between land and sea, and have been traditionally classified as forests occupying overwash islands, coastal fringes, riverine areas, and intertidal basins; scrub forests and other unique settings do occur, especially in relation to the dominance of carbonate (Lugo and Snedaker, 1974).

Mangroves are of great importance economically to coastal inhabitants and ecologically as an integral part of the coastal zone throughout low latitudes and are a prime source of wood for fuel and construction; chemicals for traditional medicine; food; breeding grounds and nursery sites for many terrestrial and marine organisms; sites of accumulation of sediments, carbon, nutrients, and contaminants; as well as offering some protection from erosion and from catastrophic events, such as tsunami and cyclones (Alongi, 2008).

Global distribution, biogeography, and losses

Mangrove forests grow throughout the low latitudes with their global distribution best circumscribed by major ocean currents and the 20 °C isotherm of seawater in winter (Figure 1). Indonesia, Australia, Brazil, and Nigeria accommodate about 43 % of the world's mangroves, which comprise a total global area of about 138,000 km^2 (Spalding et al., 2010). There are roughly 70 true mangrove species in 40 genera in 25 families (25 species belong to two families, the Avicenniaceae and Rhizophoraceae) that occur only in these tidal forests, plus a loosely defined group of mangrove associates that also occur in lowland rainforests, freshwater swamps, and salt marsh (Tomlinson, 1986). What is meant by the term "mangrove" is botanically ambiguous, as many different families and genera are not closely related phylogenetically. Mangroves thus represent an ecological rather than a taxonomic assemblage of woody plant species having a variety of common morphological, biochemical, physiological, and reproductive attributes that enable them to inhabit saline soils waterlogged by comparatively warm tidal waters.

Mangroves first appeared on the shores of the Tethys Sea, having diverged from terrestrial forbearers during the Late Cretaceous-Early Tertiary (Ricklefs et al., 2006). The number of species is greatest in the Indo-West Pacific, fostering the traditional explanation that mangroves originated in Southeast Asia and expanded eastward across the Pacific to the west coast of the Americas and westward to East Africa and then to the east and west coasts of the Atlantic. Movements of the Earth's plates were responsible for the separation of what was once a continuous global distribution; as the various plates separated, some species invariably became extinct, while others diversified regionally. Today, there is a clear separation between the Indo-West Pacific (IWP) and the Atlantic-Caribbean-East Pacific (ACEP) biogeographic regions, as cold waters prevent contact and dispersal between the southern Indian and Atlantic Oceans. Thus, there are six subregions (Western Americas, Eastern Americas, Western Africa, Eastern Africa, Indo-Malesia, Australasia) that fall into the distinct IWP and ACEP regions (Figure 1). Constraints on their global distribution include temperature, rainfall, and human impacts (Duke et al., 1998). Although quite variable geographically, both the number of mangrove species and their total area decline with increasing latitude or decreasing rainfall or both. A poleward expansion of mangroves on at least five continents in relation to the poleward extension of temperature thresholds has occurred concurrent with sea-level rise (Saintilan et al., 2014).

Despite their importance along tropical and subtropical coasts, mangroves are disappearing at an alarming rate due to clearing for coastal development, for aquaculture, and for timber and fuel production (Daru et al., 2013; Polidoro et al., 2010). A global loss rate of 1–2 % has been cited (Spalding et al., 2010), but some areas experience little loss, while others are losing a greater percentage of total area. Approximately 15 % of the world's mangrove species are at a high threat of extinction (Polidoro et al., 2010), especially along the Atlantic and Pacific coasts of Central America, where up to 40 % of mangrove species present are currently at risk. The most landward forests are most often the most threatened with clearing.

Adaptations

Mangroves have many morphological, reproductive, and physiological attributes that make them well adapted to salty soils. These specializations include aerial roots, viviparous embryos, and tidal dispersal of propagules. Mangroves exhibit a number of different strategies to deal with salt, such as salt avoidance and regulation coupled with mechanisms of tolerance; resistance strategies include exclusion, extrusion, storage, succulence, compartmentalization, and osmoregulation (Popp, 1995). Salt tolerance varies greatly among species and there are wide species-specific variations in growth responses. Some species such as *Avicennia marina* do not grow in freshwater and may be obligate halophytes, whereas other species grow well in freshwater and do not have an obligatory need for more than trace amounts of salt. The presence of salt can constrain water relations, as a positive water balance and photosynthesis can only be maintained if the potentials in the plant are lower than in the soil; in high-salinity soils, maintaining water balance presents a problem of trying to take up essential inorganic ions to maintain osmotic balance while trying to avoid adverse effects of high ionic levels in the cytoplasm (Lovelock and Ball, 2002).

Because the metabolic cost of maintaining water balance is high, mangroves display a number of features to minimize water loss, including low transpiration rates and sclerophylly, expressed as mangrove leaves being thick-walled, usually with a multilayered epidermis covered by a thick, waxy, lamellar cuticle that helps to minimize evaporation. On the lower leaf surface, there is usually a dense field of hairs (e.g., *Avicennia*, *Pemphis*) or scales (e.g., *Camptostemon*) that cover salt glands and stomata to reduce water loss from these openings. Sunken stomata, waxy coatings, a thick cuticle, and widespread, cutinized, and sclerenchymatous cells are xenic characteristics for dealing with a physiologically dry environment (Saenger, 2002). Low transpiration rates are imposed by high salt concentrations. When salinity is lower due to high rainfall in the wet season, transpiration rates and stomatal conductance can be high. However, mangroves overall follow a very conservative water-use strategy.

Conserving water reflects a trade-off between the need for the stomata to open to maintain intercellular CO_2 concentrations and the simultaneous loss of water vapor; thus, carbon gain is balanced by some water loss. Low stomatal conductance limits such water loss but also restricts the uptake of CO_2. This dilemma results in low intercellular CO_2 concentrations, low assimilation rates, and high water-use efficiencies – the ratio of carbon assimilated to water used (Saenger, 2002). High water-use efficiency is achieved by adaptive traits such as specialized leaf and stomatal anatomy, high levels of photooxidative protection, hydraulic architecture (small vessels and dense wood), and greater carbon investment in roots than aboveground tree parts (Feller et al., 2010). Physiological stress is minimized and water-use efficiency is maximized by the ability of many species to adjust the angle of their leaves to avoid high temperatures and maximize heat loss; this adaptation also has a metabolic cost as maintaining a favorable leaf angle comes at the expense of light harvesting and assimilative capacity. Various leaf sizes have also evolved to help achieve a balance between maximizing carbon uptake, minimizing leaf temperatures, and minimizing water expenditure.

Morphological and physiological adaptations to maximize root aeration are a key feature of mangroves to deal with the problem of the lack of oxygen and the presence of potentially toxic metabolites in waterlogged saline soils. Morphological adaptations include relatively high root/shoot ratios as well as a range of aboveground root systems, such as pneumatophores (e.g., *Avicennia*, *Sonneratia*) that break the soil surface from the cable roots; stilt roots (e.g., *Rhizophora*) that branch off from the lower trunk and descent into the substrate; knee roots (e.g., *Bruguiera*) that break the surface but curve back down into the soil; buttress roots (*Xylocarpus*, *Heritiera*) that also branch off from the trunk but do so as flattened, triangular structures; and aerial roots that originate from the trunk or lower branches but usually do not reach the

Mangroves, Figure 2 An extensive monospecific stand of *Bruguiera gymnorrhiza* in northern Australia. Note the dense canopy and the absence of an understory.

soil (Figure 2). Some species possess one or more of these root types, but a few species commonly found in aerated and/or coarse-grained deposits close to the soil surface (e.g., *Exocoecaria*, *Aegialitis*) do not have such specialized systems (Tomlinson, 1986).

These various root systems provide aeration for subsurface roots and anchorage for the tree. More frequently inundated mangroves possess a greater variety of aboveground root types. Mangrove roots are composed mostly of aerenchymatous tissue, honeycombed with open gas spaces that run down the longitudinal axis (Tomlinson, 1986); the more that roots are waterlogged, the more gas space that is required for internal conduction. The presence of lenticels in most roots provides further evidence of the need for root ventilation. Gas transport by mangrove roots varies in synchrony with the tide. During tidal immersion, oxygen concentrations decline inside the roots with a concomitant reduction in gas pressure. At low tide when the roots are exposed to the atmosphere, the low gas pressure induces the flow of air back into the roots leading to a renewal of oxygen concentration. Transport of oxygen from roots is so efficient that in some genera the rhizome is surrounded by less hypoxic soils.

Waterlogging leads to a number of other physiological and metabolic changes. Soil anoxia induces mangroves to reduce water stress which in the case of shoots may lead to reduced growth rates due to the accumulation of ethylene or imbalance of gibberellin in the plant, as well as depressed stomatal conductance, photosynthesis, and oxygen transport via the roots and increased foliar sodium levels. The presence of anoxic metabolites (e.g., H_2S) can lead to root hypoxia, which can inhibit nutrient uptake, reduce tolerance to increasing salinity, and lower rates of root respiration. Mangroves demonstrate a variety of metabolic adaptations to the lack of oxygen (Saenger, 2002).

Mangroves also exhibit a range of adaptations to maximize reproductive success in a hostile environment. Pollination occurs either via wind or by animals such as birds, bats, bees, and other insects, but all mangroves disperse their seedlings by tides. All species of the family Rhizophoraceae produce viviparous seeds (propagules), which germinate precociously while still attached to the parent tree; the embryo ruptures the pericarp and can grow to considerable lengths. Some genera (e.g., *Aegialitis*, *Avicennia*, *Aegiceras*, *Lagunularia*, *Nypa*, *Pelliciera*) produce cryptoviviparous seeds in which the developing hypocotyl does not penetrate the pericarp; both vivipary and cryptovivipary incur considerable parental investment. The seeds of the remaining mangrove species do not germinate while still on the parent tree, but do pass through a resting stage before germinating. There may be some advantage of vivipary, including rapid rooting, prolonged nutrient uptake, and development to maximize the chances of reproductive success, and development of buoyancy. There is little evidence of long-distance dispersal of propagules (Hogarth, 2007), but buoyancy, large size, and food storage may confer some local advantage in maximizing survival that is patchy in time and space; large propagules survive longer and grow better as new recruits than small ones.

Most reproductive activity coincides with the wet tropical summer months, a time conducive to rapid growth as well as dispersal immediately after summer storms and monsoons. Viviparous seeds may possess adequate food reserves due to their extended development while still on the parent tree and are capable of relatively rapid establishment, but there is a trade-off in that such comparatively large seedlings attract a number of predators – to the extent that seed predation can play an important role in determining recruitment success and species composition and community structure of forests.

Forest structure and dynamics

Patterns of recruitment and of the forest structure that eventually develops are the end result of complex interactions among propagule survivability, environmental factors, including climate, and phenology (Feller et al., 2010). The interplay between extreme trait plasticity and specialized adaptations characterizes mangroves and their environment. Mangrove traits are highly plastic in relation to salinity, nutrient availability, and other environmental drivers such as temperature, light, sea-level rise, and the extent of tidal inundation (Krauss et al., 2008). Such plasticity of plant traits can result in forests of variable species composition, age, and community structure.

The apparent zonation of one or a few species across the intertidal seascape has long been regarded as a conspicuous feature of mangroves (Figure 2). Zonation is an oversimplification as any classification is subjective; some forests conform quite well to such categorization, while others defy any classification. Many drivers have been suggested as the causative agent for these tidal gradients, including geomorphological controls; physiological adaptation to physical gradients, especially salinity; tidal sorting of propagules; differential predation on propagules; and interspecific competition (Smith, 1992; Crase et al., 2013). Some or many of these factors can drive intertidal zonation and can vary within a coastal region and even within an estuary. Environmental gradients and species responses are thus very complex. No one factor regulates zonation where it occurs. It has even been suggested that the process is random with the first species and successive ones present eventually determining community composition (Ellison et al., 2000).

The establishment of seedlings through to mature stages is a complex process for forested ecosystems, with terrestrial forests showing peak structural complexity in middle age with a slow decline toward senescence. Disturbance and recovery in terrestrial forests play a key role in maintaining forest diversity and community structure. Present theory indicates that when species die, they are replaced by fast-growing species that are poor competitors, leading over time to eventual replacement of these pioneers by a succession of superior competitors – mostly to monopolize light – until an equilibrium is achieved in climax and postclimax sequences (Odum, 1981). Mangrove forests, in contrast, appear to have species and communities with more pioneer-stage than mature-stage characteristics, including light-demanding seedlings, competition for light, dispersal by tides rather than by biota, long propagule dormancy and viability, dependence on seed reserves, continuous production of numerous propagules, early reproductive age, uniform crown shape, prolonged flowering period, poor species richness, no or little canopy stratification, few climbers, and few epiphytes (Smith, 1992).

Changes in forest structure and composition occur within a milieu of natural disturbance to equilibrium or steady-state conditions, as all ecosystems are subject to a variety of disturbances that are a driving force in facilitating adaptive change. The timescale in which mangroves recover from disturbance depends in part on the intensity, duration, and scale of the disturbance. Recovery may follow classic large gap-phase dynamics whereby enhanced recruitment rates are matched by greater mortality following gap formation as the forest returns to a closed canopy state; the primary factor limiting recruitment in gaps is light availability.

Trends in recovery are not stochastic, but the early sequences of species replacement are greatly determined by the species present at initial recovery (Souza et al., 2007). With increasing forest age, tree densities decline but individual trees become larger due to self-thinning; with fewer but larger trees, aboveground biomass increases with forest age. Long-term changes in mangrove forest structure have rarely been examined, but a few studies suggest that mangroves are a mosaic of patches of different stand ages if there is a high frequency of gaps (Berger et al., 2006). If gaps are absent or few, there are still transitory variations in what are otherwise zonal or monospecific forests; intermediate disturbances are unlikely to culminate in a classic climax or postclimax community (Lugo, 1980). As stated by Alongi (2008, p. 5), “stand composition and structure in mangrove forests are the new result of a complex interplay of physiological tolerances and competitive interactions leading to a mosaic of interrupted or arrested successional sequences in response to physical/chemical gradients and to changes in geomorphology.” That is, if a forest remains relatively undisturbed for long time periods relative to individual life spans or if a primary forest is being established, mangroves can undergo a series of successional stages similar to those that are undergone in terrestrial forests. However, in most coastal regions, intertidal areas are frequently disturbed by natural (typhoons, seasonal monsoons) or anthropogenic (wood harvesting, pollution) forces, so mangroves are often a patchwork of interrupted successional stages, as are most ecosystems undergoing ecological succession under stress (Odum, 1981).

Mangrove forests, partly for these reasons, have low plant diversity, have a relatively simple architecture, and rarely have a significant understory, but they do have a variety of features that help make them resilient to disturbance. These characteristics include (1) a large reservoir of belowground nutrient pools that serve to replenish nutrient losses; (2) rapid rates of plant-microbial-soil cycling of carbon and nutrients that facilitate retention of these elements; (3) complex and highly efficient biotic controls, such as high rates of nutrient-use and water-use efficiency; and (4) positive and negative feedbacks that provide malleability to help dampen variations in recovery. Their simple architecture can lead to rapid recovery or rehabilitation post-disturbance, as there is redundancy of keystone species.

Fauna and trophic structure

Trees and bacteria constitute the bulk of forest biomass, but many other organisms originating from adjacent terrestrial and marine environments are found in mangroves (Macnae, 1968). Birds, bats, monkeys, tigers, insects, fish, amphibians, reptiles, and a rich fauna of estuarine and marine plankton and benthic invertebrates spend all or part of their life cycle in the forest canopy, soils, and tidal waterways (Kathiresen and Bingham, 2001; Nagelkerken et al., 2008). These populations and communities overlap, as mangroves are ecotones having a high level of connectivity with both terrestrial and marine ecosystems (Feller et al., 2010).

The most functionally conspicuous organisms in mangroves are sesarmid and grapsid crabs, being keystone engineers in many forests (Cannicci et al., 2008; Nagelkerken et al., 2008). The significance of crabs as biological drivers of forest structure and function was recognized in the late 1970s and early 1980s (Kathiresen and Bingham, 2001). Sesarmid crabs are very important in shaping mangrove structure and functioning, especially in Old World mangroves, while ocypodid crabs play the same role in New World mangroves. Crabs process a large proportion of algal and leaf biomass and are ecosystem engineers in their ability to bioturbate and aerate mangrove deposits and, along with other predators, consume and bury large quantities of propagules (Kristensen, 2008). A number of models have been proposed to explain the mode of crab control via propagule predation on forest structure: (1) the dominance-predation model, which states that there is an inverse relationship between predation rates of different species in relation to its dominance in the forest; (2) the canopy-gap-mediated model, which postulates that predation can be more intense under closed canopies than in relatively large gaps; (3) the flooding regime model, which suggests that the level of propagule predation is inversely related to inundation time (the more time the forest floor is flooded, the less time crabs have to prey on propagules); and (4) the spatiotemporal biocomplexity model, which hypothesizes that while low water or high water conditions are a key driver in initial plant establishment or failure, other factors related to the dry or wet conditions (e.g., salinity, drought, soil texture) come into play, so forest patch structure may be the result of differences in environmental drivers. Regardless of causative mechanism(s), the net result of propagule predation is reduced completion among saplings.

Mangrove structure and function are shaped by a variety of other organisms. Gastropods, for example, do so by consuming large volumes of mangrove material, such as litter, algae, and wood, and wood-boring isopods help to facilitate fungal decomposition of wood. Similarly, herbivorous insects such as ants, moths, and caterpillars bore into wood as well as consume flowers, fruits, seeds, and leaves. Ant-plant interactions can play a key role in regulating vegetation structure and function (Cannicci et al., 2008).

Above the substratum, mangrove roots are often overgrown by epibionts such as tunicates, sponges, algae, and bivalves, while the forest floor and canopy – visited by birds, insects, bats, rats, and even monkeys – are fauna-rich. Both the canopy and epibiotic communities are very diverse with close associations between tree and animals; some of the interactions are highly complex (Ellison and Farnsworth, 2001). The functional significance of many of the organisms, especially the vertebrates, is largely unknown. Root epibionts are known to be highly diverse and an attractant for a wide assortment of invertebrates, as well as having an important role in nitrogen transformation processes (Ellison et al., 1996).

Mangrove plankton and nekton are, like their benthic and canopy-living counterparts, key players in the flow of materials and energy in mangrove ecosystems. In both soils and tidal waters, a large proportion of organic matter and energy flow is funneled through a highly diverse, actively growing, “microbial loop or hub” consisting of Archaea, bacteria, protists, and viruses and subsequently transferred to higher consumers such as zooplankton and fish; metabolic by-products such as respired CO_2 and wastes are similarly transferred and integrated into various biogeochemical cycles that help to sustain life (Figure 3). Mangrove microbes are highly abundant and productive, fuelled by new and recycled DOM and inorganic nutrients. Indeed, the first link in the “microbial loop or hub” is the uptake of exudates from phytoplankton cells and cell contents released during “sloppy feeding” by microzooplankton, so there is a strong link between microbial and phytoplankton productivity. Trophic relationships within and between microbial assemblages are virtually unknown, but presumably intense, as protists such as amoebae and flagellates are voracious consumers of bacteria and are known to graze heavily on bacterioplankton (Lee and Bong, 2007). In contrast, phytoplankton communities in mangrove waters are thought to be species-poor due to inhibitory effects of high concentrations of soluble tannins and other polyphenolics. Phytoplankton abundance and productivity range widely in mangroves, usually in relation to light availability and flushing rates of waterways; stagnant or polluted waters are common throughout Asia, for example, with high rates of primary production in these virtual nutrient “soups.”

Zooplankton communities are the crucial link between microbes (to which some of the tiniest zooplankters belong), penaeid shrimps, and zooplanktivorous fish. The main factor controlling zooplankton abundance and species composition is the seasonal change in salinity, with the onset of the monsoon season the prime stimulus

Mangroves, Figure 3 An example of *Rhizophora apiculata* with both large stilt roots and extensive roots descending from lower branches; photo taken in a mixed forest in lower Sumatra, Indonesia.

for shifts in composition and abundance. Mangrove zooplankton communities consist of four components: (1) a stenohaline marine group that penetrates into the estuary mouth, (2) a euryhaline group that penetrates further up estuary, (3) a true estuarine component, and (4) a freshwater group in the upper tidal reaches (Kathiresen and Bingham, 2001). The most conspicuous group are members of the cyclopoid copepod family Oithonidae that may have a selective advantage of small size to avoid prey and a number of strategies to maximize growth and reproduction. Larger zooplankters are an important trophic link to fish but play an equally crucial role as structuring agents of microzooplankton, the latter accounting for as much as 75 % of potential phytoplankton production.

Penaeid prawns and fish have received the most attention among mangrove biota due to their role in commercial and recreational fishing. Prawn species of the genera *Penaeus*, *Metapenaeus*, *Parapeneopsis*, and *Macrobrachium* are the primary fishery targets, and great effort has focused on their life history strategies, distribution, abundance, degree of habitat dependence, and catch per unit effort (Manson et al., 2005). Shrimps function as mid-level and top omnivores, regulating the abundance of smaller plankton and nekton and spending their post-larval and juvenile stages in mangrove estuaries until emigrating offshore where they spawn in the wet season. Annual shrimp production is more a reflection of catch per unit effort than a true reflection of their productivity, but rates (13–756 kg ha^{-1} $year^{-1}$) are well within the range of values measured in estuaries and nearshore habitats worldwide (Alongi, 2009).

Fish life cycles are similarly well known, with species richness of permanent and temporary residents being a function of salinity, microhabitat diversity, tides, water depth and clarity, coastal water currents, and proximity to seagrass beds and coral reefs (Faunce and Serafy, 2006). The number of species in any given mangrove estuary can range from <10 to nearly 200, with a tendency for more species in larger estuaries; density and biomass estimates are similarly variable, ranging from 1 to 160 fish m^{-2} and 0.4–29 g m^{-2} and generally greater than in temperate estuaries (Blaber, 2002). Mangrove fish are grouped into five feeding guilds – herbivorous, iliophagus, zooplanktivorous, piscivorous, and benthic invertebrate feeders – but many species shift their dietary preferences as they age. Wild fish production varies greatly (17–1,000 kg ha^{-1} $year^{-1}$) worldwide (Alongi, 2009), with peak landings during the post-monsoon and summer months.

Are mangroves major nursery grounds for fish and shrimp? This idea was first articulated nearly fifty years ago (Heald, 1969), but the links between mangroves and edible items were obviously known by indigenous communities much farther back in time. Three hypotheses have been offered to explain the connection between coastal fisheries and mangroves: (1) the food hypothesis, which suggests that mangroves offer an abundant variety of foods; (2) the refugia hypothesis, which suggests that mangroves function as a refuge from predation; and (3) the shelter hypothesis, which suggests that mangroves provide shelter from physical disturbances (Manson et al., 2005). None of these hypotheses are mutually exclusive, and perhaps all three factors may be operating at the same time in the same place. Very little direct evidence exists to explain the relationship between fishery catch and mangroves, but evidence exists to support the notion that coral reef fish use mangroves and seagrasses as essential juvenile habitat (Kimirei et al., 2013). The underlying mechanisms or the cause-and-effect relationships of the connection between fishery yields and mangroves, however, remain poorly understood.

Forest production and photosynthetic performance

Mangroves are among the most productive plants in the sea, as revealed by proxy measurements of leaf and wood production (Alongi, 2009). Belowground production of roots has rarely been measured, but aboveground net primary production (AGNPP) averages 11 t DW ha^{-1} $year^{-1}$ (Alongi, 2009). This AGNPP rate compares favorably with the mean AGNPP rate for tropical terrestrial forests of 12 t DW ha^{-1} $year^{-1}$. Production rates of both mangroves and other tropical forests overlap highlighting the fact that similar ecological and physiological factors limit production of all trees. Some forests of both habitats in the dry tropics are not very productive compared with the most luxuriant forests in wet tropical regions. Mangroves, like other forests, vary in size and age and in the balance between production and respiration. Measurements of belowground primary production are lacking for all tropical forests, as are measurements of dark leaf respiration and respiration of roots and woody parts, so true estimates of total forest net primary production are sparse. Despite these constraints, mangrove AGNPP declines with increasing distance from the equator, mirroring the decline in mangrove biomass (Alongi, 2009).

The light response curves of mangrove leaves are similar to other tropical plants in which saturation is reached at 300–400 μmol photons m^{-2} s^{-1} after a steep linear increase to this threshold. Maximum CO_2 assimilation rates can often exceed 25 μmol CO_2 m^{-2} s^{-1} although most rates lie between 5 and 20 μmol CO_2 m^{-2} s^{-1} as saturation is reached at comparatively low light conditions due to low stomatal conductance and intercellular CO_2 concentrations. Rates of leaf photosynthesis decline with increasing salinity and increasing vapor pressure deficit. Despite these limitations, the median rate of mangrove leaf photosynthesis (12 μmol CO_2 m^{-2} s^{-1}) is equal to the median rate (11 μmol CO_2 m^{-2} s^{-1}) for shade-intolerant terrestrial trees.

The rate of dark leaf respiration in mangroves ranges from 0.2 to 1.4 μmol CO_2 m^{-2} s^{-1} with photosynthesis to respiration (P/R) ratios ranging from 2.1 to 11.2, which is at the upper end of the range of values for tropical terrestrial species. A few root respiration measurements have been made, mainly on *Avicennia marina* pneumatophores, with highly variable rates among all species (*Rhizophora mangle*, 0.5–6 nmol CO_2 g^{-1} root FW s^{-1}; *Avicennia marina*, 2–3 μmol CO_2 g^{-1} root FW h^{-1}). These rates are at the lower end of values for other tropical trees.

The uptake and assimilation of micro- and macronutrients play a key role in determining mangrove growth and production, as mangroves are often limited by the availability of nitrogen (N), phosphorus (P), and iron (Fe). These and many other elements are required to synthesize cells and to manufacture structural and reproductive tissue (Reef et al., 2010). The critical need for N and P has been demonstrated for a number of mangroves, with clear interactive effects among different mangrove species, nutrients, and environmental factors such as salinity, temperature, soil type, and frequency of tidal inundation (Feller et al., 2010).

The ordinarily high photosynthetic rates for mangroves drive a high requirement for nutrients; nutrient-use efficiencies and rates of nutrient resorption by leaves are correspondingly high. Differences among species in nutrient-use and resorption rates can be explained by differences in the way species allocate nutrients as well as species differences in leaf life spans and how energy and nutrients are vested in chemical defenses. With such a high requirement for nutrients, mangroves have evolved a number of conservation mechanisms (in addition to high resorption efficiencies), including large reservoirs of dead roots belowground, tidal export of predominantly refractory matter, and highly efficient nutrient cycles (Reef et al., 2010).

Phytoplankton and algae living on the forest floor and as epiphytes on aboveground tree parts are additional sources of fixed carbon in mangroves. Algal production in mangrove waters and under the canopy is light limited, and while algal productivity can be high, it is usually dwarfed by tree production (Alongi, 2009).

Nutrient cycling and sources for secondary consumers

The cycling of essential elements such as N in mangrove ecosystems is highly complex and internally regulated by the trees and their interrelationships with soil, water, and microbiota – most of whom are responsible for the bulk of nutrient transformations and recycling processes. Concentrations of dissolved and particulate N and P are low in tropical waters and mangrove soils, but cycle quickly, to the extent that residence times for many nutrient pools are on the order of minutes to hours. Nutrients are transformed, taken up, and assimilated by the trees and other biota both rapidly and efficiently, despite the fact that acquiring nutrients is not a straightforward process. In mangrove soils, for instance, Fe oxyhydroxides and metal sulfide complexes readily bind to organic nutrients, thereby limiting the amount available to the plant. Geochemical redox reactions in soils and sediments are complex and involve a wide array of elements and compounds restricting the uptake of dissolved organic and inorganic nutrients.

Conserving nutrients is advantageous, so large belowground reservoirs of dead roots and maximizing nutrient storage in the youngest tree parts are efficient retention mechanisms. Another effective conservation strategy is to increase the efficiency of various metabolic processes and the utilization of nutrient pools. Mangroves invest a large proportion of root metabolism in the uptake and assimilation of soil ammonium, which translates into a comparatively low-energy investment compared with using nitrate or possibly DON.

The soil N cycle in mangroves indicates that ammonium production (ammonification) is the dominant N transformation process, with proportionally little loss to the atmosphere via anammox (anaerobic ammonium oxidation) and denitrification (production of N_2 gas). High rates of ammonification are supported by dissolved organic nitrogen (DON) released by roots. The uptake of ammonium is fast enough compared to its production that often nearly all ammonium is taken up by tree roots. Nitrogen fixation and denitrification rates, in contrast, are highly variable and, on average, slow processes. The tight coupling between trees, microbes, and soil nutrient pools is partly a function of the interactive effects between forest age and intertidal position. However, the cycling of P and other elements is poorly known in mangroves.

The need for N in mangrove food webs has attracted much debate as early notions of animal nutrition pointed to N in mangrove detritus as the main fuel for food webs (Heald, 1969). The original paradigm was that microbes attached to detritus reduced complex indigestible vascular plant material to simple, more readily digestible forms, with subsequent microbial enrichment providing sufficient N for adequate nutrition (Ellison and Farnsworth, 2001). The current paradigm derived primarily from work using stable isotopes is that most mangrove consumers preferentially assimilate fresh benthic microalgae and macroalgae, phytoplankton, and algal detritus to meet their N requirements. Such material is nitrogen-rich and more readily digestible than mangrove plant detritus. The discovery that sesarmid and grapsid crabs are foundational ecosystem engineers led to a paradigm shift in recognizing the nutritional importance of algal foods. Crabs were once a prime example of the notion that mangrove secondary consumers met their nutritional needs by voraciously eating large amounts of N-poor mangrove litter and assimilating the associated N-rich microbial biomass. This early explanation seemed reasonable because crabs can consume nearly all standing stock of litter in some forests and paste litter fragments onto their burrow walls facilitating fungal and bacterial colonization, making the material more palpable and nutritious over time (Kristensen, 2008). However, mangrove litter is high in tannins and too poor in nitrogen to sustain adequate nutrition. In reality, crabs, like most other secondary consumers, eat a variety of foods to maintain a balanced diet, obtaining sufficient N from supplemental consumption of animal tissue and meiofauna. Sophisticated feeding experiments have found that the crabs *Episesarma* spp. and *Perisesarma* spp., as well as penaeid shrimp larvae, are omnivorous, eating mostly mangrove detritus and lesser amounts of roots, algae, animal tissue, and surface microbial biofilms (Nordhaus et al., 2011; Gatune et al., 2012).

Mangrove-associated fish and zooplankton also have varied diets but with a preference for algal over detrital foods. An individual mangrove forest may thus have multiple food webs partitioned by dietary preferences for algae, detritus, mixed algal/detrital foods, and animal tissues such as carcasses and smaller consumers such as microzooplankton (Giarrizzo et al., 2011). The nutritional situation can be even more complicated if seagrasses and coral reefs are nearby, as fish residents feed mainly on mangrove-associated foods, but transient fishes actively forage on seagrass or reef-associated prey items (Vaslet et al., 2012). The significance of mangrove N (and other nutrients) thus depends on the location and type of habitat, relative availability of other primary producers, species dietary preferences, and universal need to maintain a balanced diet.

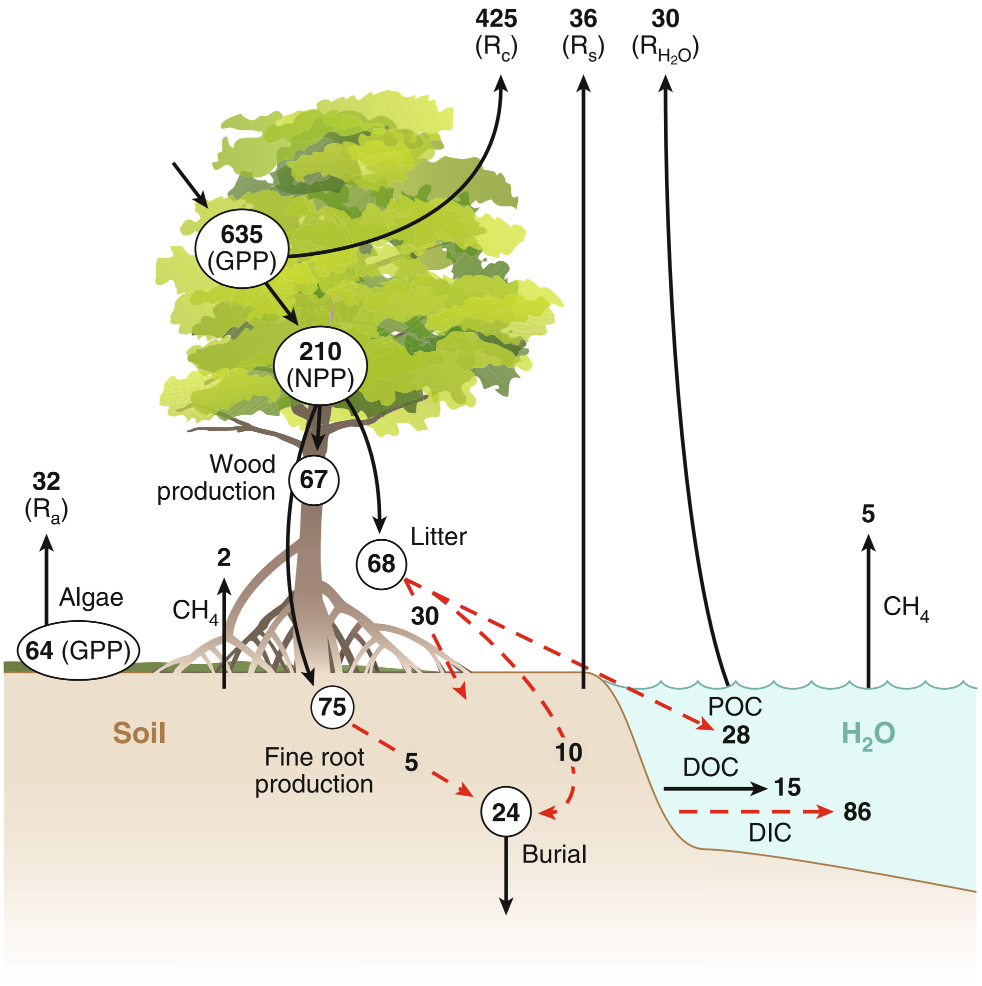
Ecosystem processes: contribution of mangroves in global coastal ocean

Mangrove forests are structurally and functionally interlinked with adjacent terrestrial and marine ecosystems and the atmosphere, exchanging biota, air, water, soils and sediments, nutrients, and other elements, over space and time. Tides represent an energy subsidy, doing work by transporting dissolved and particulate materials, gases, and metabolic by-products between the forest and other ecosystems. The idea that the fertility of coastal wetlands such as mangroves via these exchanges supports primary and secondary production in the adjacent nearshore zone developed in the late 1960s (Heald, 1969), and this “outwelling” hypothesis generated much subsequent research into the role of mangroves in supporting coastal production. After decades of research, it is clear that the amount of material potentially available for export from mangroves is influenced by such factors as forest production, tidal range, the ratio of mangrove to watershed area, local hydrodynamics, amount of rainfall, volume of water exchange, and the extent of activities of crabs and other biota (Alongi, 2009).

Most mangroves export nutrients, but some do not. Globally, mangroves export an average of 28 T g C year^{-1} of particulate organic carbon (POC), or about 10–11 % of particulate terrestrial carbon export to the global coastal ocean (Alongi, 2014). Mangroves also export large quantities of dissolved organic carbon (15 T g C year^{-1}) or dissolved inorganic carbon (86 T g C year^{-1}) to the coastal ocean, with most of the latter originating from subsurface advection of interstitial water within the forest floor. Thus, nearly 75 % of the total C exported from mangroves originates from respiration by microbes and other mangrove biota in soils and tidal waters.

A nitrogen mass balance model of the world’s mangroves indicates that (1) 2687 G g N year^{-1} is required to sustain global mangrove NPP; (2) N burial is about 25 % of total N input into an “average” mangrove forest; (3) about 15 % of total N input to mangrove soils is denitrified; (4) nitrogen fixation accounts for only about 5 % of total N input although nitrogen fixation on aboveground tree parts and in deep root systems has not been adequately measured; (5) production of roots and litter accounts for 40 % and 50 % of mangrove NPP, respectively; (6) tidal losses equate to about 55 % of N input; (7) denitrification and N_2O effluxes account for <10 % of total N losses; and (8) despite proportionally large tidal losses, the global flux of N in mangroves is roughly in balance (Alongi, 2013).

The balance of carbon between photosynthetic gains by autotrophs and respiratory losses from all biota, reflected in the exchange between ecosystems, atmosphere, and adjacent ecosystems, is called the net ecosystem



Alongi DM. 2014.
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Mangroves, Figure 4 Budget of the major pathways of carbon flow (T g C year^{-1}) through the world's mangrove forests. *Solid black arrows* represent empirically derived mean values, and *dashed arrows* represent either mean values estimated indirectly (by difference) or pathways suggested from the recent literature. Abbreviations: *DIC* dissolved inorganic carbon, *DOC* dissolved organic carbon, *GPP* gross primary production, *NPP* net primary production, *POC* particulate organic carbon, R_a algal respiration, R_c canopy respiration, R_s soil respiration, $_{RH2O}$ waterway respiration (From Alongi, 2014). Reprinted with permission from the annual Review of Marine Science, Volume 6 copyright 2014 by Annual Reviews, http://annualreviews.org.

production (NEP). NEP varies widely among mangrove ecosystems, but on average, mangroves produce and store more carbon than they lose. A number of carbon mass balance estimates exist for the global expanse of mangroves (Bouillon et al., 2007; Alongi, 2009), with the most recent budget (Figure 4) showing a number of key features of mangrove ecosystems: (1) two-thirds of forest gross primary production is lost as canopy respiration; (2) NPP is nearly evenly allocated to wood, litter, and fine root production; (3) roughly 60 % of carbon buried in soils is likely derived from litter and dead roots with the remainder originating from adjacent upland and marine ecosystems; (4) roughly 50 % of leaf litter is exported by tides with the other half utilized within mangroves; and (5) nearly all carbon (90 %) lost to adjacent coastal waters and to the atmosphere is derived from respiration. NEP of the world's mangroves equates to 90 T g C year^{-1}, a value that compares favorably with NEP for the world's coral reefs (84 T g C year^{-1}), but much less than NEP for salt marshes (475 T g C year^{-1}), seagrasses (533 T g C year^{-1}), and macroalgae (2,221 T g C year^{-1}). Mangrove ecosystems are net autotrophic, with a GPP/R ratio of 1.15, with the remaining fixed mangrove carbon stored in vegetation and soil and, to a much lesser extent, is lost to a variety of human uses. Mangroves occupy 0.5 % of coastal ocean area but account for 8 % of coastal respiration, 7 % of coastal GPP, and 3 % of coastal NEP. Mangroves account for approximately 10–15 % of total carbon sequestration in the coastal ocean. These budgets show that cycling of carbon and other elements is in rough balance in mangrove ecosystems, despite living in a harsh, tropical environment.

Summary and conclusions

Mangroves occupy only about 0.5 % of the world's coastal area but contribute disproportionately in myriad ways to the ecology and economy of tropical and subtropical coastal zones and their inhabitants. Mangrove forest biomass and production are equivalent to tropical lowland forests. Mangroves have evolved many morphological, reproductive, and physiological traits for life in waterlogged saline soils, including aerial roots, viviparous embryos, sclerophylly, low assimilation rates, high root/shoot ratios, and high water- and nutrient-use efficiencies. The forest structure is structurally simple compared with their distant terrestrial relatives, often lacking an understory and having comparatively low tree diversity; species richness is greatest in the Indo-West Pacific supporting the notion that this is also the location of their origin during the Late Cretaceous-Early Tertiary. Tidal gradients in species composition are frequently expressed in relation to combinations of tidal gradients in salinity, frequency of tidal inundation, seed predation, competition, and other drivers, the complex interplay of which leads to forest mosaics of interrupted successional sequences. Trees and bacteria account for most forest biomass, but rich pelagic, arboreal, and benthic food webs consist of both terrestrial and marine flora and fauna. Sesarmid and grapsid crabs are foundational ecosystem engineers, although microbes drive carbon and biogeochemical cycles. Mangroves are among the most productive plants in the sea, being highly efficient users of essential micro- and macronutrients.

While statistics for most countries are lacking, many mangrove forests are no longer pristine, even in the most remote locations. In the face of forecasted rises in sea level, the pressures on mangroves worldwide are expected to increase for the foreseeable future.

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Cross-references

Coastal Wetlands
Estuarine Connectivity
Forested Wetland Habitat
Species Zonation
Tidal Flat
Trophic Dynamics

MARINE/FRESHWATER MIXING

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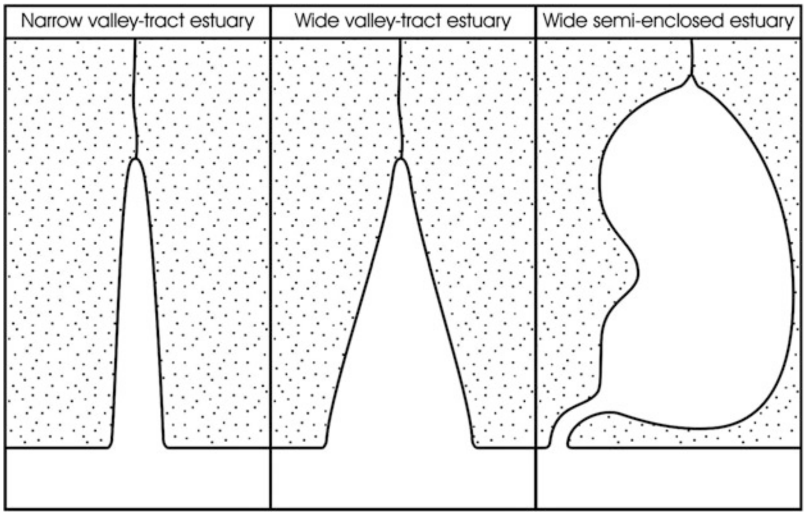
Definition

In an estuarine context, the mixing of marine and freshwater refers to the dynamics of interchange and dilution by freshwater delivered mainly by riverine influx and of marine water delivered by tides, wind-driven currents, and wave action.

The estuarine environment: a zone of mixing between seawater and freshwater

The essence of an estuary is that it is a river-to-marine transitional environment where marine salinity is measurably diluted by (riverine) freshwater in a valley tract, an inlet, a coastal lagoon, or an embayment, producing a salinity gradient from the river to the sea (Cameron and Pritchard, 1963; Pritchard, 1967). As such, the estuarine environment is the zone of mixing between freshwater derived from river sources and marine water from the sea. In terms of hydrochemistry, biota, and processes, there is a riverine component toward the landward part of an estuary and marine component seaward (Day, 1981; Dalrymple et al., 1992; Allen and Posamentier, 1993; Semeniuk et al., 2000; Semeniuk and Wurm, 2000).

Landward parts of estuaries with perennially flowing rivers can remain freshwater throughout the year. Landward parts of estuaries with seasonally flowing rivers fluctuate between freshwater and brackish or between freshwater and marine salinity. At the other extreme, the marine environment generally remains at seawater salinity but, with seasonally strongly flowing rivers, fluctuates between marine and brackish or even freshwater (where freshwater riverine plumes enter the sea). Over the period of a year, a season, or a week, depending on flow rates and volume of delivery of river water and tidal flows, the central estuarine basin can remain perennially brackish or may fluctuate between freshwater and marine with periods of brackish water or fluctuate between brackish and marine. In areas of high evaporation and minimal river influx, the headwaters or even main waters of an estuary may become hypersaline.



Marine/Freshwater Mixing, Figure 1 Idealized illustration showing three geometric forms of an estuary that will have influence on the types of marine and freshwater mixing: (1) narrow valley-tract type where there is confinement of flow and currents, (2) wide valley-tract type where there is less confinement of flow and currents, and (3) wide semi-enclosed estuary where there can be development of complex mixing and complexity of estuarine landforms.

The salinity of an estuary and the mixing within it are also affected by processes and factors beyond just the seesawing of seawater influx and riverine water outflow and can involve hydrodynamics driven by wind, waves, seiching, and evaporation, among others, and different sources of water. For instance, the shape and size of an estuary and whether it can be subject to complications of a variety of mixing processes can have a large influence on mixing types (Figures 1 and 2).

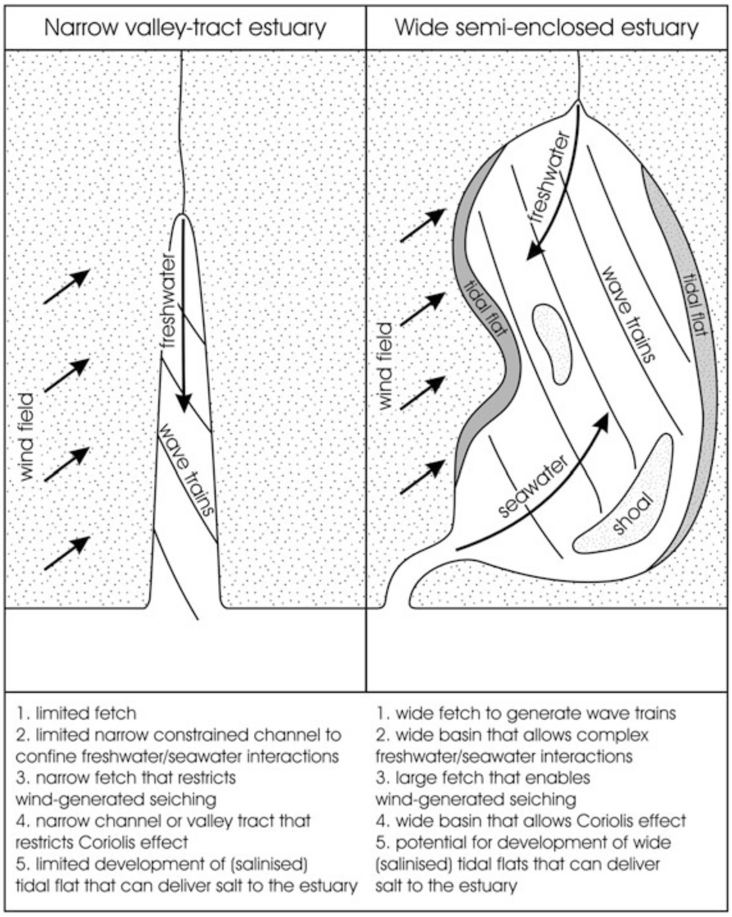
The sources and types of water entering an estuary are (Figures 3 and 4) (1) marine water, (2) other saline waters deriving from the shoreline, (3) freshwater deriving from river flow, (4) freshwater deriving from groundwater seepage/discharge, (5) freshwater deriving from rainfall, and (6) saline water formed by evaporation of the estuarine water body.

Marine water is the major source of saline water and is delivered to the estuary by tides and wind-driven currents. Other saline waters derive from the shoreline. The most important sources are high-tidal flats which can become salinized on exposure during neap tides, or during the periods of prevailing lower water. When these tidal flats are flooded on high tides or flushed by surface freshwater flows deriving from the uplands surrounding the estuary, saline water is delivered to the estuarine shoreline. The other dominant water type, freshwater, is delivered by river flow. With perennial and strong river flow, freshwater can penetrate far into the estuary and even enter the sea; with seasonal river flow, or moderate river volumes, freshwater penetrates only a limited distance into the estuary. Freshwater flows to the estuary from shoreline groundwater seepage (Semeniuk, 1983; Cresswell, 2000) and subaqueous discharges (Semeniuk et al., 2011) are relatively minor, but the timing and location of such flows can be significant. Direct rainfall, as a source of freshwater into the estuary, is particularly important in areas of high precipitation. In arid or highly evaporative climates, a specific source of water is produced by evaporation by which salts are concentrated in surface waters.

Processes that drive mixing

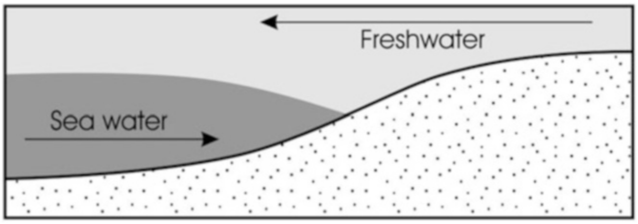
There are a number of processes, operating at different scales, which combine to deliver and/or to mix the six water types (Figures 3, 4, 5, and 6). At the largest scale, there is:

1. Tide-driven marine water influx
2. River flow (river currents) into saline water
3. Shoreline freshwater seepage
4. Sub-estuarine freshwater seepage into shallow water
5. Freshwater runoff from salinized tidal flats
6. Tidal flooding, dissolution, and ebb-tidal flow of salinized tidal flats
7. Tidal creek discharge
8. Saline water density-driven flows
9. Wind-driven currents
10. Wind current inducing Ekman flow and vertical mixing
11. Wind-induced surge
12. Vertical currents driven by wind-induced surges
13. Seiching
14. Wind-induced surface water cooling and the resulting vertical circulation/convection
15. Temperature-determined density-driven flows
16. Turbid water density-driven flows
17. Evaporation leading to denser saline surface water, followed by sinking



Narrow valley-tract estuary	Wide semi-enclosed estuary
1. limited fetch	1. wide fetch to generate wave trains
2. limited narrow constrained channel to confine freshwater/seawater interactions	2. wide basin that allows complex freshwater/seawater interactions
3. narrow fetch that restricts wind-generated seiching	3. large fetch that enables wind-generated seiching
4. narrow channel or valley tract that restricts Coriolis effect	4. wide basin that allows Coriolis effect
5. limited development of (salinised) tidal flat that can deliver salt to the estuary	5. potential for development of wide (salinised) tidal flats that can deliver salt to the estuary

Marine/Freshwater Mixing, Figure 2 Idealized estuaries of the narrow valley-tract type and the wide semi-enclosed type showing contrasting hydrodynamics and estuarine landforms that will result in and influence types of marine/freshwater mixing.

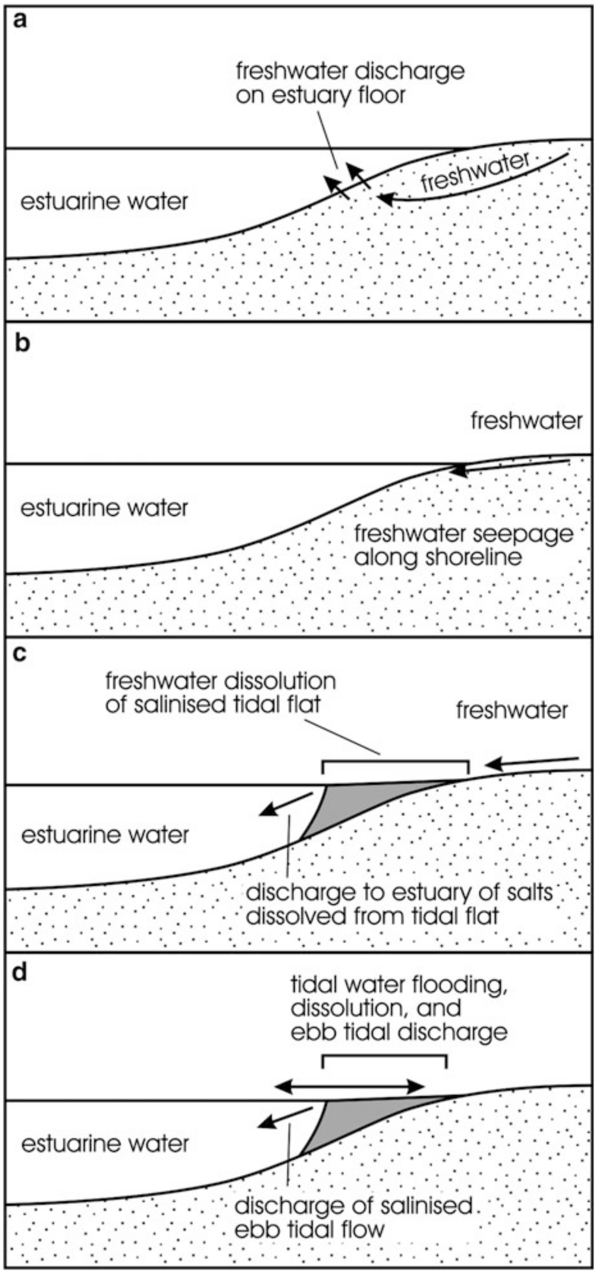


Marine/Freshwater Mixing, Figure 3 Simplified diagram showing the two extreme and major water-body sources in an estuary, viz., seawater and freshwater, entering the estuary from two different directions. The less dense freshwater flows out over the seawater.

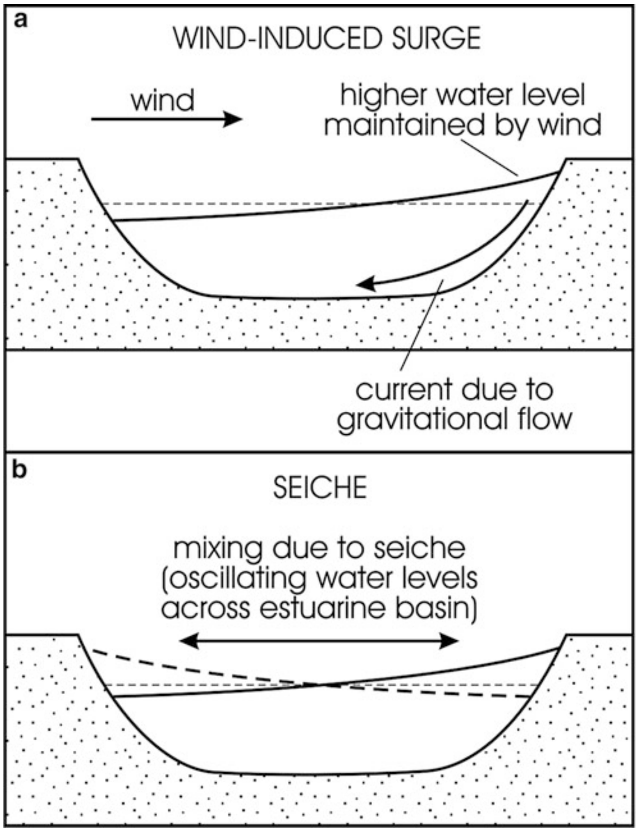
18. Bathymetry-induced and plan-geometry-induced eddying
19. Horizontal mixing driven by interaction of flood tidal water and outflowing freshwater
20. Mixing by Coriolis force

At the finer scales, across the saltwater/freshwater interface, there is (1) entrainment, (2) turbulence, and (3) wave-driven (wave orbital) vertical mixing. Intermittent and infrequent events such as storms and cyclones can also induce mixing especially at the seaward portion of an estuary.

Not all the processes that drive mixing are present in all estuaries. The extent that a process is effective is dependent on the size of the estuary (in that some of the mixing mechanisms operate best in large estuaries), bathymetry of the estuary (in that complex bathymetry can result in complex mixing and eddying), plan geometry of the estuary (in that complex planar geometry results in complexity in mixing), tidal range, rainfall in the catchment area of the contributing river(s) (which will determine the volume and consistency of freshwater delivery), the wind patterns and strengths (which determine the importance of wind-driven circulation and of intra-estuarine wave trains), and the climate (that influences the amount of evaporation that will salinize the estuary or its tidal flats).



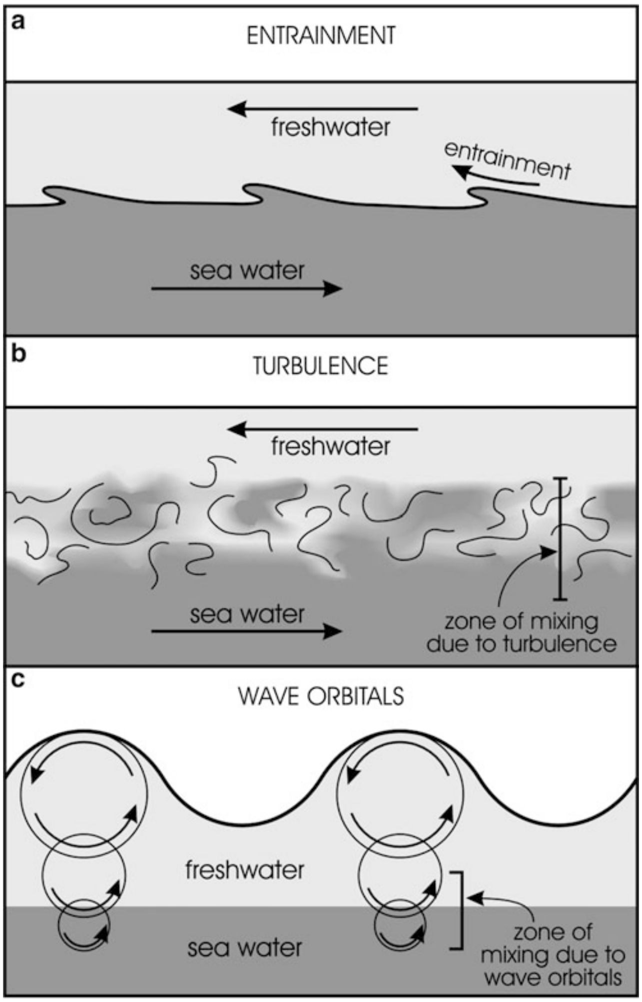
Marine/Freshwater Mixing, Figure 4 Additional large-scale locations and mechanisms that provide freshwater or saline water to an estuary. In these four examples, salinity stratification is not shown, but the extent of salinity structure will complicate the resultant mixing. (**a**) Subterranean discharge of freshwater subaqueously into the floor of an estuary; this seepage would dilute the saline lower depths of the estuarine water. (**b**) Surface seepage of freshwater into the shore of an estuary. (**c**) Surface seepage of freshwater from the shore and across a salinized tidal flat (if present) contributing saline water to the estuary. (**d**) Flood and ebb-tidal flow across a salinized tidal flat contributing saline water to the estuary.



Marine/Freshwater Mixing, Figure 5 Additional large-scale mechanisms that mix marine water and freshwater in an estuary. (**a**) Wind-induced elevated water level (wind-induced surge) (Ratkovich and Ivanova, 2001) which, when sustained, also may result in a gravitationally induced counter currents along the estuary floor. (**b**) Seiche (oscillating water levels), resulting from wind-induced elevated water level (wind-induced surge) followed by cessation of the wind shear.

Some of the hydrodynamic processes that result in changes of elevation to the surface estuarine water (specifically tidal propagation, river freshwater inflow, and the Coriolis acceleration) are subsumed under the term “barotropic pressure” (Feireisl, 2004). Barotropic pressure gradients drive flow and hence mixing. Horizontal changes in density that can result in density-driven flows are subsumed under the term “baroclinic pressure gradient” (Li et al., 1998; Simons et al., 2010).

The mixing of the various water bodies in estuaries is more easily viewed from the perspectives of major and subsidiary or minor contributing processes. The major processes involved in mixing the water types are tides, river flow, and wind. Here, for tides and river flow, the effectiveness of mixing is directly determined by the comparative magnitudes of tidal and river fluxes and their interactions and of the duration and strength of river flow. Daily tidal influx delivers marine water that commonly



Marine/Freshwater Mixing, Figure 6 Small-scale mechanisms that mix marine water and freshwater in an estuary. (**a**) Entrainment along the seawater/freshwater interface as the two water bodies flow in opposite directions. (**b**) Turbulence along the seawater/freshwater interface as the two water bodies flow in opposite directions. (**c**) Wave orbitals that induce mixing across the seawater/freshwater interface.

penetrates the estuary as a salt wedge under a freshwater layer. Alternate flooding and ebbing of the tide, and penetration into and under the freshwater, causes the saltwater/freshwater interface to alternate and change level resulting in mixing of the two water bodies.

The next major process is river flow, and, depending on the strength of the river, freshwater is injected into the estuary to varying degrees. At one extreme it may be injected as far as the seaward mouth of the estuary, rendering the entire estuary brackish or fresh. With strong river flow, and tidal ranges of lower magnitudes, freshwater flows over a deeper layer of more dense seawater.

The relationship of tidal magnitude and river flow and the resulting estuarine mixing and estuarine salinity structure is treated descriptively and mathematically by Haas (1977), Geyer and Farmer (1989), and MacCready (1999).

Wind is the third major process involved in mixing the water types within an estuary. It is particularly important where estuaries are microtidal and the tides are less effective in driving the mixing of water bodies. With adequate fetch and strong winds, major horizontal circulation patterns may be established which can crosscut salinity plumes and salinity gradients resulting in cross-estuary mixing. Strong winds can also result in surges which, with elevated water levels along a shore, result in gravitationally induced return-flow basal currents. If derived from low-salinity near-surface water, these flows descend as anomalous lower-salinity plumes into deeper water. Wind-induced surges can also lead to seiching (Jackson, 1833; Ratkovich and Ivanova, 2001; Luettich et al., 2002), which results in cross-estuary mixing. Depending on the size and depth of the estuary, horizontal wind currents can induce Ekman flow and hence also vertical mixing (Gross, 1972). Finally, wind can result in wind-induced surface water cooling and gravitational sinking of the cooler surface layer with a resulting vertical circulation/convection.

Other relatively more minor processes involved in mixing of estuarine water include shoreline freshwater seepage (where freshwater discharges from the uplands along the shore) and sub-estuarine or subaqueous freshwater seepage into shallow water (where freshwater discharges from the uplands into aquifers which convey it into the subaqueous environment). If the subaqueous freshwater discharge from the estuary floor enters more saline water, the buoyancy of the freshwater creates a plume to the surface of the estuarine water body and contributes to local mixing.

Where salinized water from tidal flats and tidal creeks enters estuarine water of lower salinity, the local saline water density-driven flows can result in local mixing. Sources of local salinized water can arise in several ways. Estuarine tidal flats can become salinized during exposure during neap tides or low spring tides, and freshwater runoff from the uplands, crossing these salinized tidal flats, becomes salinized and delivers salty water to the estuary. Similarly, tidal flooding, dissolution, and ebb-tidal flow of tidal waters across salinized tidal flats deliver salt-enriched water to the estuary on the ebb tide. Tidal creeks draining and ebbing from salinized tidal flats can deliver salt-enriched water in plumes to the estuary.

Wind cooling of estuarine surface waters can result in temperature-determined density-driven flows where surface waters, sufficiently cooled, gravitationally descend to warmer parts of the estuarine water body resulting in the mixing of water bodies. Mixing also is effected by density-driven flows resulting where turbid freshwater, laden with suspended mud, is delivered by a river to the estuarine basin and where evaporation leads to denser saline surface water which sinks into the less dense water.

Horizontal mixing can take place where strongly inflowing flood tidal waters enter a large estuarine basin through a relatively narrow inlet to meet a stream of strongly outflowing freshwater emanating from a similarly narrow river tract. The interaction is a large-scale circulation pattern. Some degree of horizontal mixing is also effected in large estuarine basins by Coriolis force, which is the flow induced on currents due to the rotation of the Earth (Gross, 1972; Dyer, 1973).

Finally, smaller-scale mixing results from eddying induced by bathymetry complexity and by plan geometry of the estuarine basin. This is the result of river flows, tidal flows, and wind-driven currents occurring in estuaries of complex planar topography or with complex cross-sectional and longitudinal bathymetry.

At finer scales, along the saltwater/freshwater interface, mixing is effected by the following: entrainment (where the interface is sheared and disrupted by the flowing freshwater and saline water is incorporated into the overlying water by advection) progressively leading to turbulence (where flow is rapid enough to change from laminar flow along the saltwater/freshwater interface to turbulent flow) (Wu, 1973; Kantha et al., 1977) and wave-driven vertical mixing induced by wave orbitals (Martin and McCutcheon, 1999; Qiao et al., 2004; Babanin, 2006).

The description of mixing in the text above has largely concentrated on the interaction of saline and freshwater wherein the density differences are mainly due to salt content and to some extent temperature differences. Further complications in mixing occur where density of water is determined by sediment in suspension or as bed load sediment transported along the river or estuary floor in traction. Mud suspended in freshwater surface flows and sand-saturated basal traction loads have a higher density than sediment-free freshwater. Instead of freshwater flowing out into denser seawater in the estuarine basin, many estuaries experience more unpredictable flows, due to density-determined dynamics of sand-dominated and mud-dominated current flows resulting in turbidity along the estuary floor and in the upper water column, respectively. Similarly, the dynamics of mixing become complicated where turbid river flow interacts with tidal flow that has been enriched with suspended mud or with turbid basal entrained sand.

Bates (1953) and Orton and Reading (1993) describe variability of turbidity and sediment loads and interactions with contrasting density in the receiving waters in three situations: (1) hypopycnal flows in which density of the suspended sediment flow is less than that of the receiving estuarine water body, (2) homopycnal flows in which density of the suspended sediment flow is equal to that of the receiving estuarine water body, and (3) hyperpycnal flows in which density of the suspended sediment flow is more than that of the receiving estuarine water body. The dynamics of sediment-laden river flow with its various amounts of suspension and/or traction load entering and interacting with an estuarine receiving water body of different density (ranging from fresh to brackish to marine) determine the different types mixing and, for sediment deposition, the different styles of sediment accumulation.

Thus, riverine freshwater flowing into an estuarine basin of denser brackish water or marine salinity will exhibit hypopycnal flow, with freshwater overlying the denser estuarine water. Riverine freshwater flowing into an estuarine basin of similar freshwater salinity, or turbid freshwater flowing into brackish water, will exhibit homopycnal flow, with the river water invading the estuarine water of similar density in a turbulent mixing front. At the other extreme, sediment-laden turbid riverine freshwater flowing into an estuarine basin of freshwater or weakly brackish salinity will exhibit hyperpycnal flow, with the denser sediment-laden river (composed of sediment in suspension and in traction) in a base flow under the less dense estuarine water.

Various estuaries and models of estuarine settings have been investigated empirically and mathematically to provide an understanding of the dynamics of mixing associated with bathymetrical variations and of circulation patterns that result from interacting processes. For instance, Lerczak and Geyer (2004) modeled the lateral circulation in straight, narrow, stratified estuaries to contrast them with homogeneous or with weakly stratified estuaries comparing flood-tide lateral hydrodynamics with ebb-tide hydrodynamics, and addressing the issues of along-channel and cross-channel differential advection and Coriolis forcing. They determined that in straight narrow stratified estuaries, lateral circulation during flood tides is about four times as strong as during ebb tides, the flood–ebb-tide asymmetry being due to a feedback between lateral circulation and the along-channel tidal currents, as well as to time-varying stratification over a tidal cycle. The modeling showed that as the stratification increased, the lateral circulation was reduced because of the adverse pressure gradient setup by isopycnals being tilted by the lateral flow itself.

Armi and Farmer (1986) explored the difference in hydrologic exchange between an estuary with a submerged barrier and one with a contraction (or constriction). They showed that exchange over a submerged barrier or sill (such as an inundated moraine) is fundamentally different to that through a contraction. Sill crests primarily affect the deeper layer into which the sill projects and only indirectly control the surface layer, consequently resulting in asymmetrical flows. Where both a sill and a contraction are present, the contraction affects flow at the sill crest only if it occurs between the sill and the source of denser water.

Chen and Sanford (2009) explored the effects of axial wind influences on salinity stratification and longitudinal transport of saline water in partially mixed straight estuaries. They showed that wind straining along the channel salinity gradient is an important control on stratification. While stratification is always reduced by up-estuary wind, it shows an increase-then-decrease transition when down-estuary wind stress increases. Such transition is a result of the competition between wind straining and direct wind mixing.

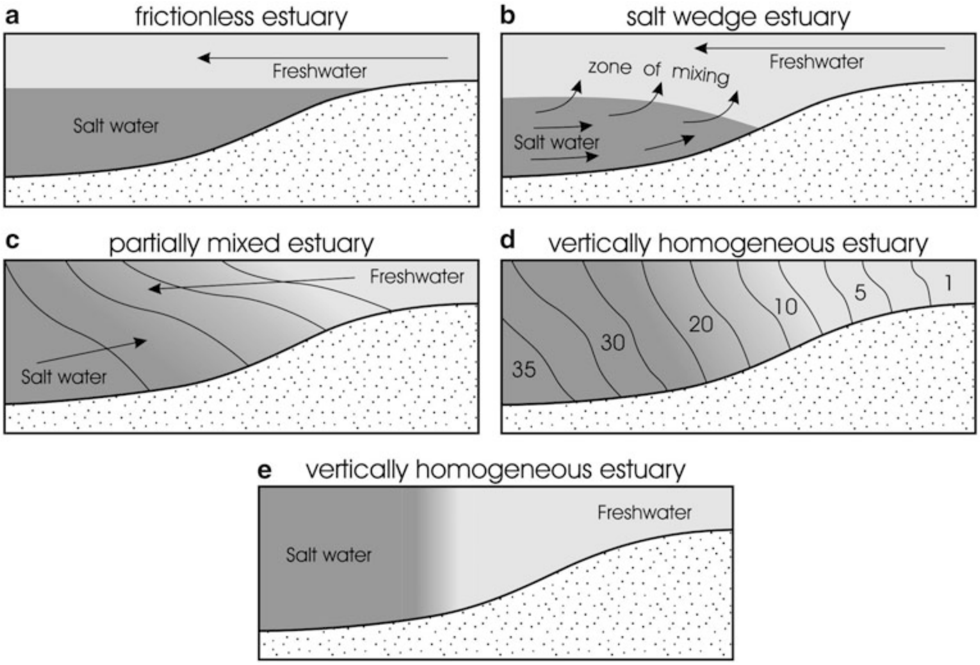
The salinity structure of estuaries

The hydrochemical (salinity) structure of an estuary from river to sea varies in response to the interaction of freshwater flow, friction, and tidal mixing. The salinity structure can be categorized as (Figure 7): (1) highly stratified (salt-wedge type), (2) moderately stratified, or (3) vertically homogeneous (Pritchard, 1955; Cameron and Pritchard, 1963; Dyer, 1973; Knox, 1986).

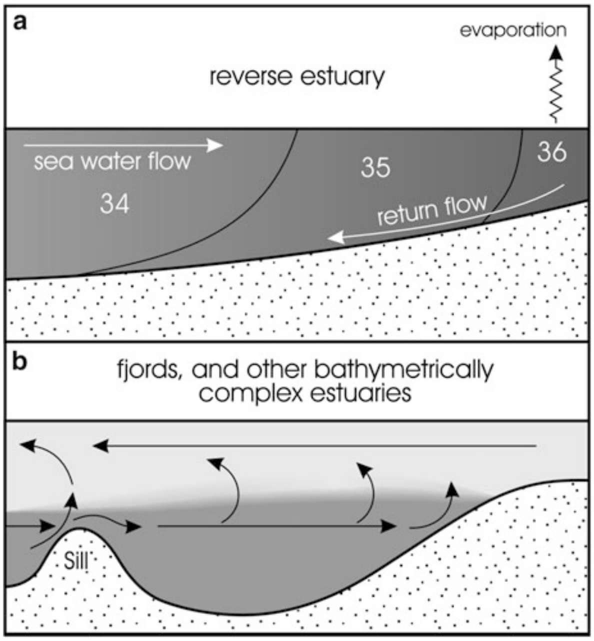
Examples of estuarine salinity structure for South African estuaries are provided by Schumann et al. (1999) and for selected estuaries around the world by Dyer (1973). A highly stratified structure results where, in the absence of friction, seawater penetrates as a wedge-shaped, deep layer tapering toward the river mouth to a distance inversely dependent on river flow (Knox, 1986) and freshwater as a surface layer penetrates far into the estuarine environment. With friction and shear along the salinity interface, the slope of the salt wedge is steepened. With increased tidal flow, mixing of water bodies occurs at all depths to form a moderately stratified estuary. Here, with freshwater at the landward part of the estuary and marine water to the seaward, the vertical mixing by river flow and tidal dynamics results in the salinity of both upper and lower layers increasing seaward, with the deeper water at any part of the estuary exceeding the surface salinity. With marked tidal forcing, the vertical mixing is complete, and the estuary becomes vertically homogeneous, dominated by freshwater and brackish water toward the river and by marine water seaward. While there generally is an emphasis on the interplay and balance of tidal forcing and freshwater flow to explain the salinity structure of an estuary varying from highly stratified, moderately stratified, or vertically homogeneous, the vertical mixing also can result or be amplified by wind circulation, wave action, and eddying due to bathymetric complexity.

Topographic/bathymetric, hydraulic, and evaporative complications to mixing patterns

Many valley-tract, simple estuaries are used as models to identify processes and to obtain principles of water mixing. However, estuaries manifest a large range of morphology, bathymetry, hydrodynamic variation, and hydrochemical variation and can be quite complex in terms of salinity structure and its temporal variation



Marine/Freshwater Mixing, Figure 7 The large-scale salinity structure in estuaries (Modified from Knox, 1986). The various marine/freshwater relationships are graded, progressing from a frictionless estuary (where freshwater overlies seawater), to a salt-wedge estuary, to a partially mixed estuary, to a vertically homogeneous estuary, reflecting the sequence from a distinct horizontal to inclined saltwater/freshwater interface with a seawater intrusion to vertically mixed salinity profiles that still maintain the marine to landward freshwater environments.



Marine/Freshwater Mixing, Figure 8 Variations and complications on the large-scale salinity structure in estuaries. (**a**) Reverse estuary where the upper estuary, which normally would be freshwater, has been salinized by evaporation. (**b**) Longitudinal section through a fjord, or other bathymetrically complex estuaries, where there is potential through topographic/bathymetric complexity for trapping or for eddying of saline water at depth thus complicating a simple salt-wedge structure.

weekly, seasonally, and inter-annually. For estuaries with perennial river flow, the salinity structure may remain throughout the year, but for estuaries with seasonal river flow the estuary may change from highly stratified, to moderately stratified, to vertically homogeneous, or even inverse, as the seasons progress (Figure 8). In other situations, bifurcating estuaries and irregularly shaped estuaries are examples, where competing interactions between tide and river flow become complicated by topographic and bathymetric variability with the potential for horizontal eddying or local trapping of pockets of water, retarding of flow and exchange. Irregularly shaped estuaries, where winds generate currents and wave trains and the wind, wind waves, and wind-induced currents are oriented with respect to the irregularity in plan form, become particularly complex in terms of mixing. Complexity in mixing also results where estuaries have multiple freshwater entry sites contrasting with the simple ocean-to-river salinity transition that is present in a narrow single valley-tract estuary with only one freshwater entry site. Large estuaries and large and deep estuaries also contrast with small estuaries in mixing styles in that there is scope in the larger systems for multiple mixing processes to occur or to be more effective (e.g., wind, seiching, Coriolis force). Finally, bathymetric complexity such as irregular bathymetry and inundated rock bars, sedimentary bars, or terminal moraines (in former glaciated areas) perturbate the basal saline layers of invading seawater and disrupt the idealized salt-wedge geometry of a stratified estuary (Figure 8).

Some case studies

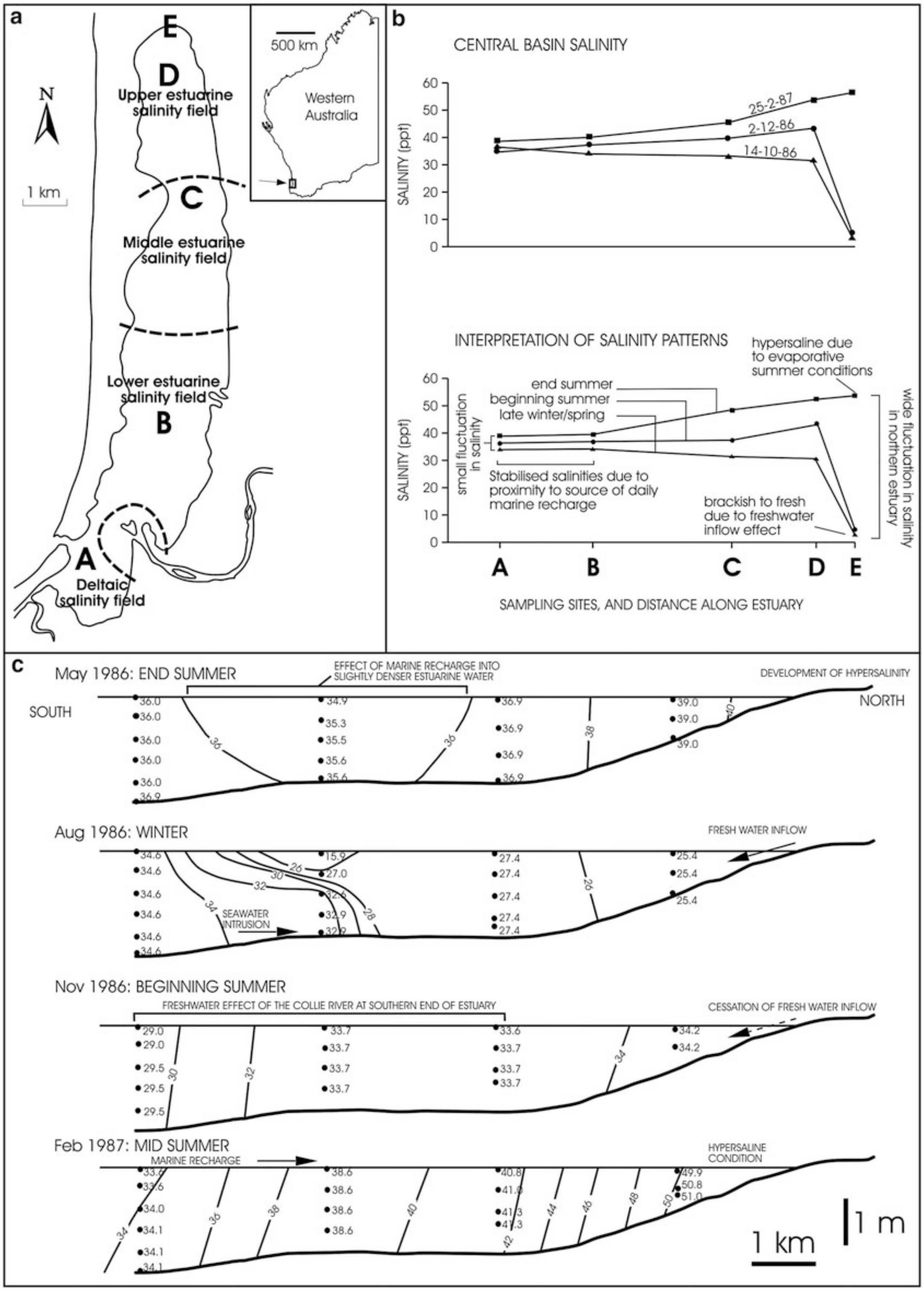
The styles of mixing and the various salinity structures in estuaries are best explained and illustrated by a range of case study examples deriving from localities in various climate and oceanographic settings. In the Mobile estuary, Gulf of Mexico, for instance, the salinity structure is related to strength of wind (in generating waves) and to strength of river flow (Schroeder et al., 1990). The relative strengths of river discharge and wind stress change the estuary from highly stratified to nearly homogeneous.

In the macrotidal estuaries of the French Atlantic coast, during the fortnightly neap-spring cycle of tidal amplitudes, the ratio of river flow to tidal volume changes considerably. This in turn results in large variations in mixing, and as such, the estuaries can change from relatively well mixed during spring tides to partially mixed, or even well stratified, during neap tides (Allen et al., 1980).

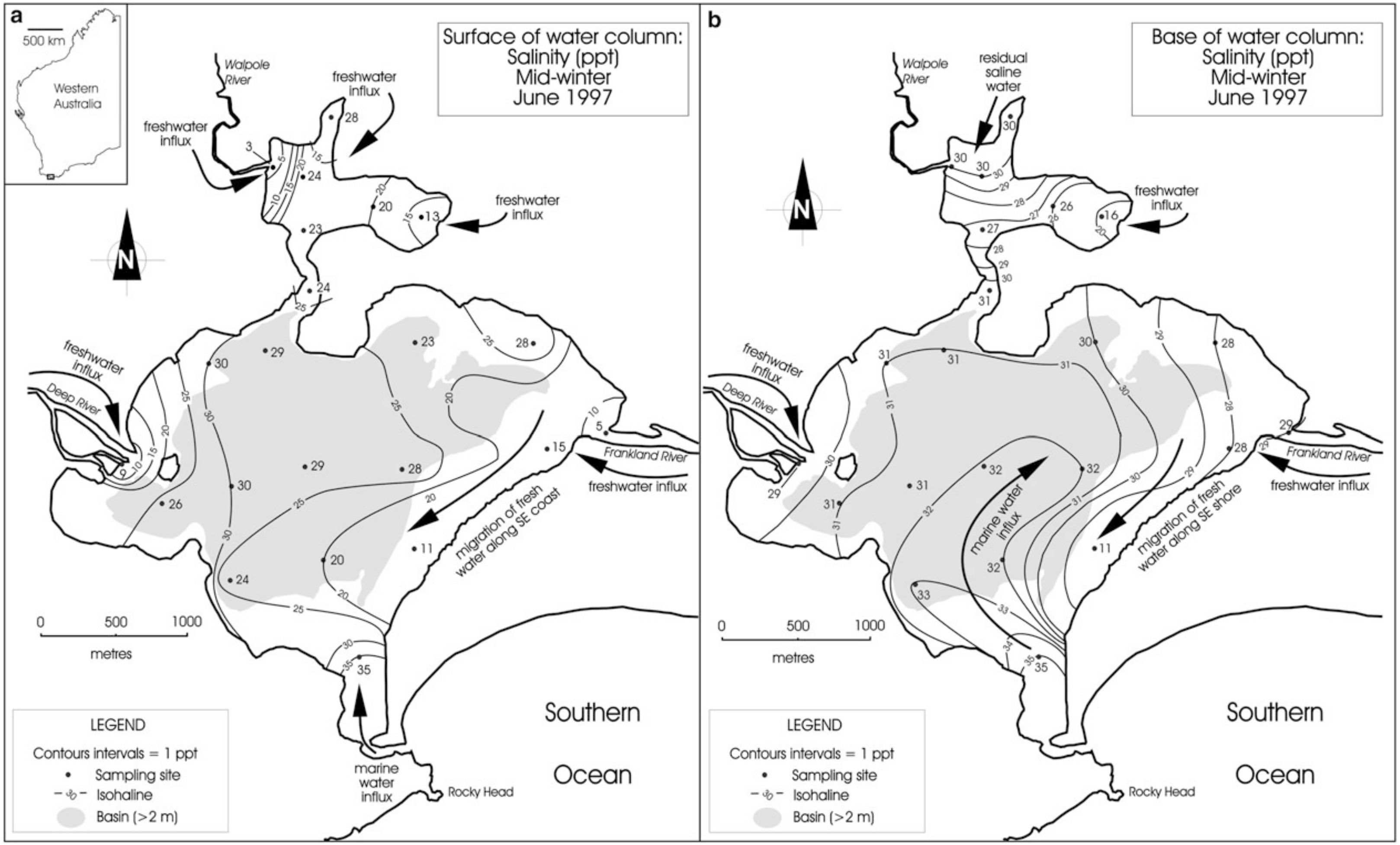
In Chesapeake Bay, the mean flow is basically a two-layered, nontidal circulation with large wind-driven velocity fluctuation and large, atmospherically induced exchange between the estuary and coastal ocean showing the importance of wind on upstream saline intrusions (Wang, 1979). Related to the velocity changes, the salinity distribution in the estuary had large variations, comparable to its seasonal change, and estuarine water responded to longitudinal (local) wind and coastal (nonlocal) Ekman flux. The response was barotropic in the lower estuary, and baroclinic (frictional) in the upper estuary. The difference in response characteristic appears to be due to the counter-effects of the near-surface wind stress shear and the depth-independent surface slope.

Eastern Long Island Sound, North America, shows the effects of bathymetry, oscillating barotropic forcing, and vertical mixing on density stratification and on the exchange between estuarine and oceanic waters (Valle-Levinson and Wilson, 1994). Intra-tidal fluctuations in density stratification appear to have been produced by both hydraulic effects and tidally induced vertical mixing, causing a fundamental change in the nature of the flow from weak to strong barotropic forcing. Vertical mixing tends to decrease exchange and to break the hydraulic control.

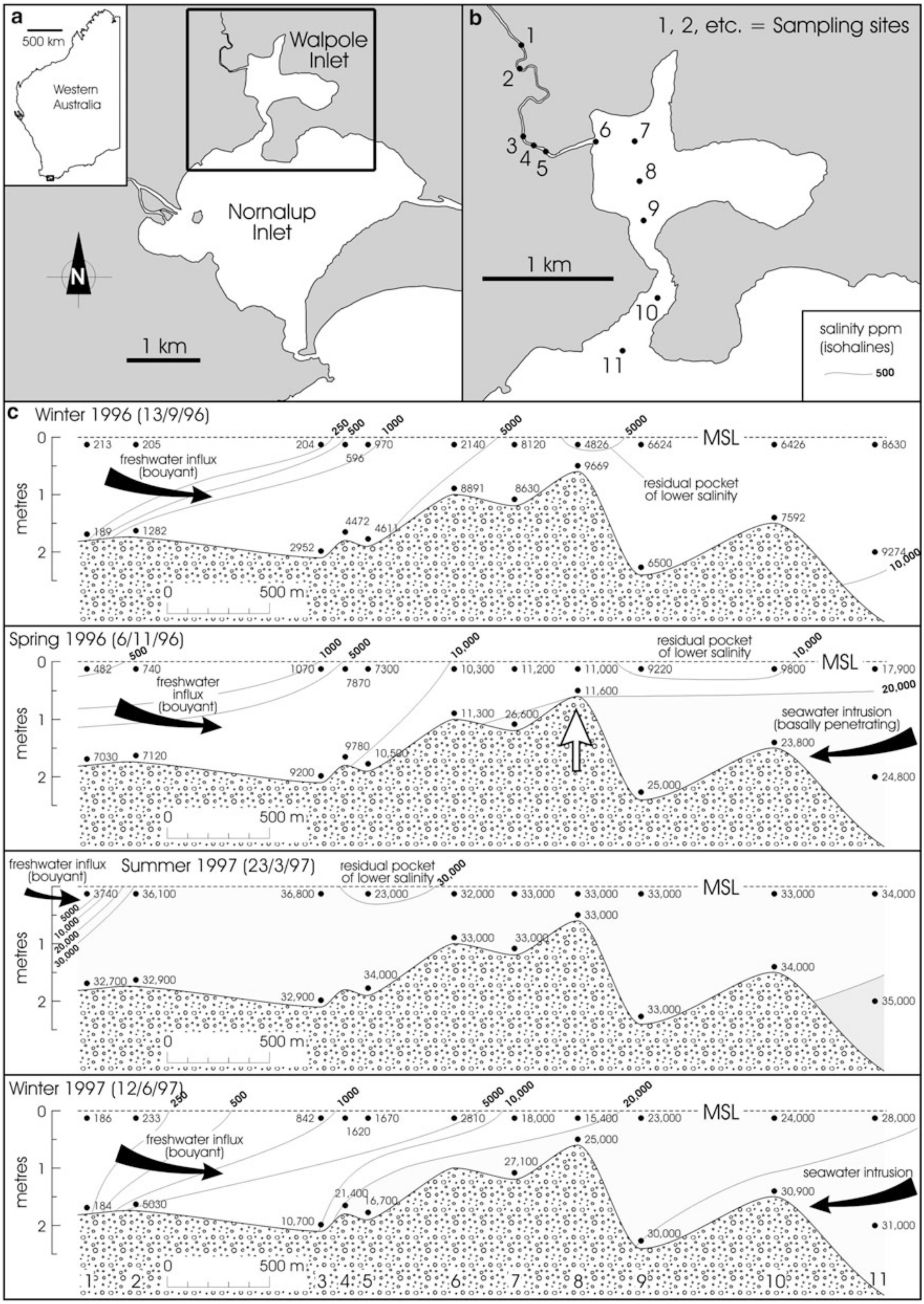
Leschenault Inlet estuary is an estuarine lagoon with river inlets to its south and freshwater seepage and runoff to its north. The estuary is wind dominated, and microtidal. During summer, with evaporation, its northern part salinizes. The studies of salinity structure show intra-annual salinity-structural changes (Semeniuk et al., 2000; Wurm and Semeniuk, 2000). The key features evident in Figure 9 are: (1) the homogeneous salinity structure for most of the year, (2) the brackish water in the northern



Marine/Freshwater Mixing, Figure 9 Salinity structure in the Leschenault Inlet estuary (After Semeniuk et al., 2000). (**a**) Location of the Leschenault Inlet estuary. (**b**) Salinity gradient from northern (*upper*) estuary to southern (*lower*) estuary over the seasons. (**c**) Salinity structure from northern estuary to southern estuary over the seasons showing (*a*) development of seawater wedge, (*b*) freshwater influx in the upper estuary in winter, (*c*) hypersalinity in the upper estuary in summer, (*d*) vertically homogeneous salinity structure in summer, and (*e*) occurrence of pockets of residual water bodies.



Marine/Freshwater Mixing, Figure 10 Salinity structure and circulation patterns in the Walpole-Nornalup Inlet estuary in midwinter during the time of major riverine flows (After Semeniuk et al., 2011). (**a**) Surface water patterns evident from the estuarine salinity. The five freshwater sources producing freshwater plumes are evident, with the Frankland River producing a major plume that is being deflected clockwise. The marine influx is evident to a small extent at the estuary outlet. (**b**) Basal water patterns evident from the estuarine salinity. Each of five freshwater sources are still evident as plumes into the estuary, with the Frankland River producing a major plume that is being deflected clockwise. The marine influx also is clearly evident as a major plume into the estuary, moving clockwise in concert with the freshwater plume from the Frankland River.



Marine/Freshwater Mixing, Figure 11 (Continued) estuary that salinizes to hypersaline in summer, (3) the saltwater intrusion in winter, and (4) the locally residual lens of seawater in slightly denser estuarine water.

The Walpole-Nornalup Inlet estuary in southern Western Australia provides a case study of salinity distribution and circulation patterns in winter and changes in the marine-to-river salinity structure over the seasons (Semeniuk et al., 2011). The estuary is microtidal and set in a humid climate with high rainfall and three rivers draining into the estuarine basin. The estuary is composed of a small northern basin (Walpole Inlet) and a larger southern basin (Nornalup Inlet). The estuary connects to the Southern Ocean via a permanently open narrow channel where tidal exchange takes place. Figure 10 shows that during midwinter in 1997, the surface waters of the northern basin of Walpole Inlet had a salinity slightly less than Nornalup Inlet. There are three areas of freshwater influx corresponding to the rivers and creeks, but with more marked freshwater dilution of the residual marine water from 35 to 30 ppt and the zone of marine water influx (with incursion of water of 35 ppt salinity). The freshwater tongue emanating from the Frankland River is close to the southeastern shore of Nornalup Inlet. The basal waters in Walpole Inlet are well mixed, saline at 33 ppt, and slightly diluted by freshwater. Similarly, the basal waters in Nornalup Inlet generally are well mixed and saline at 34 ppt, slightly diluted by freshwater. The minor influx of freshwater diluting the marine waters is evident at the mouths of the Deep and Frankland Rivers. However, the map of basal-water salinity shows a body of marine water of salinity 35 ppt, deriving from the tidal exchange channel, entering Nornalup Inlet as a curved tongue. The map also shows a tongue of freshwater emanating from the Frankland River, remaining close to the southeastern shore of Nornalup Inlet. That this freshwater is in basal parts of the water column suggests that it is relatively dense and carrying suspended mud (to create a dense, freshwater body). The geometry of flow, with the invading tongue of marine water and the efflux of freshwater, is interesting in that the marine water enters Nornalup Inlet and moves clockwise, while the freshwater from the Frankland River forms a plume, also moving clockwise. In part this is due to Coriolis Force to deflect the freshwater plume from the mouth of the Frankland River toward the south (to the left), but it is also due to the horizontal mixing driven by interaction of flood tidal water and outflowing freshwater, the former deflecting the outflowing freshwater to the right.

Figure 11 shows the longitudinal changes in salinity structure in northern Walpole Inlet over the seasons. The key points of the salinity dynamics are the buoyant freshwater plume in the spring and winter, the saltwater wedge in the winter made prominent by the prevailing fresh to brackish water of the estuary in winter, the residual pockets of less saline water in winter and spring, and the effect of a complex bathymetry in limiting and perturbating the extent of the seawater intrusion.

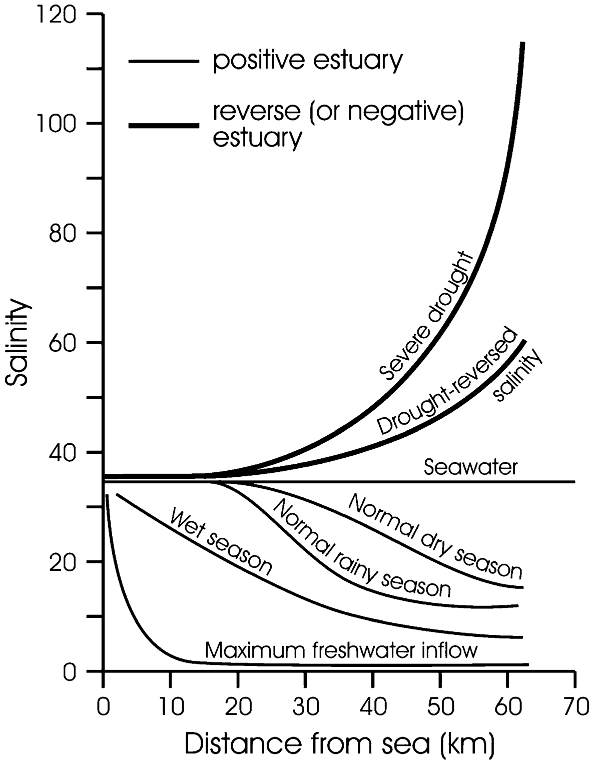
Summary

The mixing in an estuarine environment between freshwater from river sources and marine seawater is accomplished by major processes of tides, river flow, and wind and less dominant processes of seiching, eddying, gravitational flows, and Coriolis Force, among others, and at the small scale by entrainment, turbulence, and wave orbital mixing. As an outcome of the interplay of tidal intrusion, freshwater injection, evaporation, and degree of mixing, the salinity structure longitudinally along an estuary can be stratified, weakly stratified, or vertically homogenous, and the salinity gradient can be marine to freshwater from ocean to river (positive estuary) or, after evaporation during the dry season, can change to marine to hypersaline from ocean to river (negative estuary). This salinity structure can be perennial or changing between the seasons.

The salinity structure and its maintenance are determined by a number of factors such the nature of the tides, the perenniality and strength of river flow, and climate. For instance, the salinity structure of a microtidal estuary that has strong perennial freshwater flux will differ from a macrotidal estuary with strong perennial freshwater flux. Similarly, salinity structure will vary depending on whether the estuary is microtidal or macrotidal in the region of low evaporation or high evaporation and freshwater influx is seasonal. The climate influences mixing in an estuary in that there can be strong winds, high rainfall or low rainfall, and low evaporation or high evaporation.

Begg (1978, illustrated in Schumann et al., 1999) presented an elegant diagram that essentially summarizes the salinity dynamics of an estuary (Figure 12). Focused on the St Lucia Estuary in Natal, South Africa, Begg (1978) summarized the inter-annual response in salinity in a longitudinal section in response to an inter-annually varying climate from wet years to those with severe drought. While this diagram captures the variability of this single estuary inter-annually, the pattern can be applied to various estuaries around the Globe that reside in different climate settings, or any other single estuary with variable inter-annual climate.

Marine/Freshwater Mixing, Figure 11 Salinity structure in northern Walpole Inlet in the Walpole-Nornalup Inlet estuary over the seasons (After Semeniuk et al., 2011). (**a**) Location of Walpole Inlet. (**b**) Sampling sites from the river middle tract, through Walpole Inlet, and into northern Nornalup Inlet. (**c**) Salinity structure over the seasons showing (*a*) buoyant freshwater plume in the spring and winter, (*b*) saltwater wedge in the winter of 1997, (*c*) residual pockets of less saline water in the winter of 1996 and spring of 1997, and (*d*) the effect of a complex bathymetry, in particular subaqueous sills in limiting and perturbating the extent of the seawater intrusion; the *vertical white arrow* identifies one sill that limited the saline water intrusion during the spring of 1996.



Marine/Freshwater Mixing, Figure 12 Response of the salinity gradient the St. Lucia estuary to a variable climate. This diagram captures the variable response of a single estuary in terms of its longitudinal salinity to inter-annual variation in climate. The beauty of the diagram is that the pattern can be applied to various estuaries in different climate settings, or any single estuary with variable inter-annual climate (Diagram after Begg, 1978, illustrated in Schumann et al., 1999).

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Cross-references

Estuarine Circulation
Evaporation and Transpiration
Residual Circulation
Tidal Flat Salinity Gradient
Tidal Hydrodynamics
Well-Mixed Estuary

MARSH DROWNING

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Synonyms

Marsh conversion to open water; Marsh submergence; Permanent marsh inundation

Definition

Marsh drowning is the large-scale death of tidal marsh vegetation and conversion to open water habitat due to a loss of marsh surface elevation relative to the tidal frame. This process can occur as a result of sea-level rise, land-surface subsidence (due to oil, gas, or groundwater withdrawal), and/or compression of deep sediments (Brinson, 2006; Reddy and DeLaune, 2008). The ability of a marsh to keep pace with sea-level rise and land-surface subsidence is based on the buildup or vertical accretion of organic material and inorganic sediments through time on the marsh surface. If the rate of vertical accretion is not fast enough in the face of sea-level rise and/or land-surface subsidence, a marsh slowly becomes submerged and is eventually lost due to permanent flooding. This has been a long-standing problem in coastal marshes in the Mississippi River deltaic plain of Louisiana, USA (Reddy and DeLaune, 2008), but is also of increasing concern globally due to the predicted acceleration in the rate of sea-level rise.

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Cross-references

Marsh Islands
Salt Marsh Accretion
Saltmarshes

MARSH ISLANDS

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Definition

Marsh islands are coarse-grained ridges along a salt marsh margin formed by storm deposition through contemporary tidal inlets.

Description

Originally documented along the North Carolina coast (Cleary et al., 1979), inlet-associated marsh islands have been recognized in a number of back-barrier and barred estuary settings around the world. These linear ridges rise from 0.5 to >1 m above the vegetated marsh surface and mark the locations of active or former tidal inlets (Figure 1).

Marsh islands typically occur along mixed-energy barrier coasts, where a salt marsh platform is located in proximity to flood-tidal shoals. Recent studies of these features in South Carolina, Massachusetts, and Maine indicate that the stratigraphy of the islands consists of a fining-upward, shell-rich sandy sequence deposited through one or several stages of sediment influx by flood-tidal currents and storms waves. Subsequent aeolian reworking and varying stages of vegetation development characterize more mature landforms. This sequence typically has a sharp

Marsh Islands, Figure 1 Historical marsh islands behind North Inlet, South Carolina.

contact with underlying salt marsh peat, with relict islands undergoing burial by marsh around their periphery. Migrating historical inlets often show a string of progressively younger marsh islands in the direction of inlet movement. In the event of inlet closure, these back-barrier landforms, which are often younger than most parts of the fronting barrier, provide important evidence of the channel location and evolution. Although described as fetch-limited barrier islands by some authors, most marsh islands are more analogous to overtopping deposits. Boat wake wash and dredge-spoil piles may have similar morphology, but do not represent true marsh islands. Satellite image-based analysis of active inlets in contrasting geographic regions (Mid-Atlantic states, western Madagascar, and northern Siberia) shows a positive correlation between the distance from inlet throat to the marsh islands and the high-tide inlet width. The dimensions of the inlet channel, marsh topography, storm wave energy, short-term sea-level changes, and latitude-dependent factors (oyster reefs, mangrove density, ice cover) determine the ultimate geological context and preservation potential of the marsh islands. If preserved in the rock record, marsh island sand bodies will serve as localized high-permeability reservoirs within fine-grained estuarine sequences.

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Cross-references

Coastal Wetlands
Saltmarshes
Wetlands

MARSH SEDIMENT TOXICITY

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Definition

Sediments in estuaries and salt marshes in urbanized or agricultural areas can accumulate toxic chemicals. Sediments are a major repository ("sink") for many persistent contaminants, and concentrations of contaminants in sediments are often orders of magnitude greater than those in the overlying water.

Toxicity of sediments

Toxic chemicals in sediments can be harmful to benthic organisms. They can accumulate in sediments and are often transferred trophically to fishes and other animals higher in the food web. Measuring the concentrations of individual toxic chemicals in a sample of sediment provides useful information, but does not necessarily indicate how toxic it is. Higher chemical concentrations in sediments do not necessarily translate to greater toxicity because the contaminants may be so tightly bound to the sediment particles that they are not available to the organisms. Factors affecting the bioavailability of sediment contaminants include the pH, degree of oxidation of the sediments, mineralogical composition, acid-volatile sulfide (AVS), and grain size. Smaller particles have greater surface area to bind chemicals, which are thereby less available for uptake. The amount of AVS compared to the simultaneously extracted metals (SEM) can be used to evaluate the availability of metals. The SEM/AVS molar ratios are an indicator of the amount of available metals present in the sediment pore water. When SEM/AVS ratios are <1, the concentrations of metals in the sediment porewater are generally below toxic levels because of the low solubility of the metal sulfides.

The degree of toxicity of the sediments can be evaluated biologically in standardized laboratory tests using certain small benthic species (often amphipods) that are exposed for certain periods of time to dilutions of the sediments and monitored for survival (acute tests), growth, behavior, and other endpoints (chronic tests). A more ecological approach to sediment toxicity is examining the resident community in the sediments – more contaminated sediments will tend to have reduced species richness and reduced abundance of organisms. Highly contaminated sediments are also likely to be dominated by tolerant opportunistic species. Using a combination of chemical analysis, sediment toxicity tests, and benthic community evaluation – the "sediment quality triad" – provides the most extensive insights into sediment toxicity.

MASS PHYSICAL SEDIMENT PROPERTIES

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Synonyms

Engineering properties of sediments; Geotechnical sediment properties; Soil mechanics; Soil physics

Definition

The expression “mass physical sediment properties”, in engineering also known as geotechnical sediment properties, refers to the physical nature of a sediment or soil, including its behavior when exposed to external forces.

Introduction

This contribution only examines the more commonly used mass physical properties. More comprehensive technical treatments can be found in Lambe and Whitman (1969), Carver (1971), Inderbitzen (1974), Mitchell (1976), Dunn et al. (1980), Hillel (1998), and Warrick (2002). The physical properties of the sediment particles (e.g., grain size and mineralogy) are not discussed in any detail here as they are dealt with elsewhere in this encyclopedia.

The mass physical properties addressed in this contribution are the shear strength, water content, bulk density (mass concentration), void ratio, porosity, permeability, Atterberg limits, and carbonate content. Various combinations of these parameters allow a quantification or assessment of important parameters and processes, among them material standing stocks and fluxes (e.g., Iversen and Jørgensen, 1993; Bartholomä et al., 2000; Delafontaine et al., 2000), but also sediment–organism relationships in terms of, for example, biomass, species numbers or species richness per unit area, or sediment volume relative to sediment type, food resources (e.g., organic matter, particulate organic carbon, particulate organic nitrogen), or pollution levels (e.g., heavy metals, PCBs, and other toxic substances) (e.g., Pearson and Rosenberg, 1978; Jumars and Wheatcroft, 1989; Meadows and Meadows, 1991; Snelgrove and Butman, 1994; Maxted et al., 1997; Mulsow et al., 1998; Flemming and Delafontaine, 2000; Delafontaine et al., 2004).

Definition of units and associated terminology

The rules for the usage of various physical units, including their symbols and associated nomenclature, have been laid down in the International System of Units (SI) in Oceanography (e.g., UNESCO, 1985). Unfortunately these rules are even today frequently violated (see Williams, 2004). In this context, the terms *content* (mass per unit mass) and *concentration* (mass per unit volume) are often confused or used synonymously (cf. Flemming and Delafontaine, 2000). This may partly be due to the fact that the two terms are not clearly distinguished in the SI rules for units. Therefore, to avoid any misunderstandings, the units associated with material contents and concentrations are specifically defined and explained below.

Content

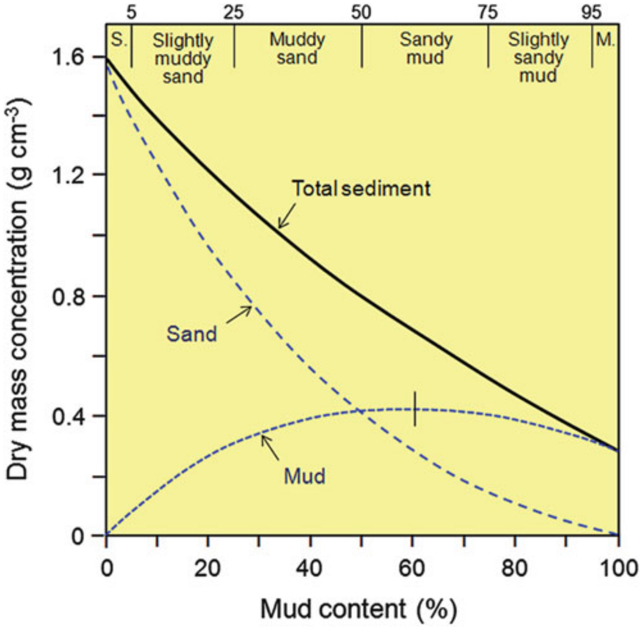
A content is a measure of the mass (or number or volume) of a substance in a unit mass of sediment and is commonly measured in g kg^{-1}, mg g^{-1}, or µg g^{-1} (or number or volume per kg, g, mg, or µg). Correspondingly, one speaks of mass content (e.g., mass of solids per unit mass of sediment), number content (e.g., particle number per unit mass of sediment), or volume content (e.g., pore space per unit mass of sediment). Contents are often expressed as percentage values of the total sediment, e.g., weight% in the case of mass contents.

Concentration

A concentration is a measure of the mass (or number or volume) in a unit volume of sediment and is commonly measured in kg m^{-3}, g dm^{-3} (or g l^{-1}, i.e., grams per liter), g cm^{-3}, or mg cm^{-3} (or number or volume per m^3, dm^3, or cm^3). Correspondingly, one speaks of mass concentration (e.g., mass of solids per unit volume of sediment), number concentration (e.g., particle number per unit volume of sediment), or volume concentration (e.g., pore space per unit volume of sediment).

Although the two measures are closely related, the relationship between the two is neither simple nor straightforward, and sometimes even counterintuitive, as illustrated in Figure 1 by the very different trends described by the total sediment (sand + mud, i.e., grain size of 2,000–63 µm and <63 µm, respectively) and the individual sand and mud concentration curves as a function of the mud content. Initially the latter two curves follow an expected trend, the mud concentration increasing as the sand concentration decreases. However, at a mud content of about 60 % (in this example), the mud concentration curve peaks and then, quite unexpectedly, also decreases with further increases in mud content. As will be explained below, the reason for this counterintuitive trend is the increasing water content of the sediment and associated changes in the network structure (fabric) of sand–mud mixtures. Note that the sediment type classification at the top of Figure 1 follows the scheme proposed by Flemming (2000).

The example in Figure 1 shows that a change in content of a sedimentary component (here mud) is automatically accompanied by a change in concentration, even though not always in a predictable or obvious way. A change in concentration, on the other hand, is not necessarily accompanied by a corresponding change in content. For example, the compaction of mud by expulsion of water (e.g., in the course of compaction) will not change the mud content per unit sediment mass, but the material concentration will increase proportionally to the decrease in volume (Figure 2a), the process being known as condensation. The opposite process is known as dilution, the concentration now decreasing, while the content again remains unchanged. Thus, material contents will only change if a solid substance is added to or removed from the sediment, e.g., the precipitation or selective dissolution of carbonate from a quartz–carbonate mixture. These mechanisms, respectively, are known as enrichment and depletion of the sediment in a particular substance (Figure 2b), the latter process being also known as impoverishment. Such changes in content are automatically accompanied by changes in concentration. The important message here is that condensation does not result in enrichment and dilution not in depletion.



Mass Physical Sediment Properties, Figure 1 Dry mass concentration as a function of mud content in intertidal sand–mud mixtures of the Wadden Sea (Modified after Flemming and Delafontaine (2000)). Note the counterintuitive trend of the mud fraction at mud contents >60 % (sandy mud and finer; sediment type classification after Flemming, 2000). The total sediment trend is determined by the dry bulk density.

Parameter definitions

Water content

The water content refers to the amount of interstitial fluid contained in a unit mass of water-saturated sediment. If the sediment is not saturated with water (e.g., in drained or partly drained intertidal sands), one would speak of the moisture or residual water content. In the case of water-saturated sediments, two types of water content measures have to be distinguished, i.e., the absolute water content and the relative water content.

The absolute water content is defined as the mass of water relative to the total mass of a water-saturated sample. This ratio is generally expressed as a percentage value (weight %), which is mathematically expressed as:

$$W_a = 100(M_w/M_t) \tag{1}$$

where W_a is the absolute water content (weight %), M_w the mass of the pore water (mg, g, or kg), and M_t the total mass of the water-saturated sample (mg, g, or kg). In this approach the percent water content is always <100 %.

The relative water content, by contrast, is defined as the mass of water relative to the mass of the dry solids in the sample. This ratio is commonly also expressed as a percentage value (weight %) and is mathematically expressed as:

$$W_r = 100(M_w/M_d) \tag{2}$$

where W_r is the relative water content (weight %), M_w the mass of the pore water, and M_d the mass of the dry solids in the sample. In contrast to the absolute water content, the relative water content can reach values >100 %, if the mass of the pore water exceeds the mass of the dry solids, as is commonly the case in soft mud. In fluid mud, for example, the relative water content can be as high as 500 %. Because relative water contents <100 % can be confused with absolute water contents, it is imperative to always indicate the type of measurement.

In environments characterized by saline waters, as is the case in most estuaries, the determination of water content by weight loss after drying needs to be corrected for the mass added by the precipitated salt. In the case of absolute water content, the procedure is mathematically expressed as:

$$W_a = 100[(M_t - M_d)/(M_d - rM_t)] \tag{3}$$

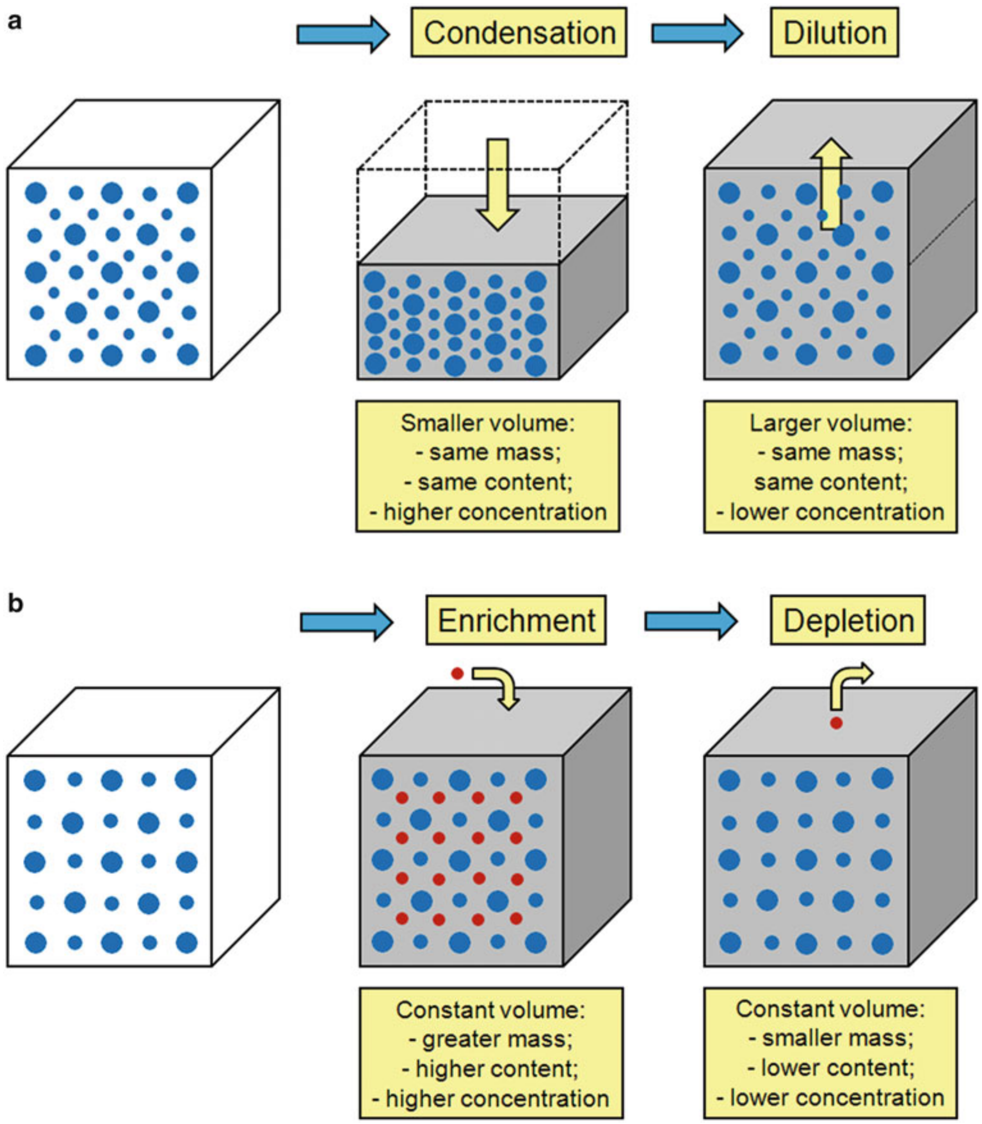
where r is the mass of salt in the pore water of the unit sediment volume, and other parameters as defined above. In the case of relative water content, the corresponding equation is:

$$W_r = 100[(M_t - M_d)(1 + r)/M_t] \tag{4}$$

with parameters as defined above.

Bulk density

The bulk density of sediment (also known as "unit density" in geotechnology) denotes the mass per unit volume of sediment. It is an indication of the degree to which the sediment is permeated by voids (pore space) relative to the volume of the solids. Bulk density is normally determined by sampling a constant volume of sediment and relating it to the measured mass of the solids. In aqueous environments, the bulk density can include the pore water, in which case we speak of wet bulk density, or it can exclude the pore water, in which case we speak of dry bulk density. An example of the trends defined by both wet and dry bulk density as a function of mud content for Wadden Sea sediments is illustrated in Figure 3a (modified after Flemming and Delafontaine, 2000; Delafontaine et al., 2004).



Mass Physical Sediment Properties, Figure 2 Definition diagram illustrating the relationship between condensation and dilution (**a**) and that between enrichment and depletion, (**b**) in unit volumes of sediment.

It should be noted, however, that in other environments the relations between bulk density and mud content may depart from the illustrated example due to regional differences in the grain-size composition of the mud fraction (silt/clay ratios) and/or changing water contents due to variable degrees of sediment compaction.

Estuarine sediments are commonly composed of terrigenous minerals such as quartz, carbonates, and clay which all have similar densities (of ca. 2.65 g cm^{-3}). Thus, if the mineral composition is not known, the average grain density (ρ_s) of the dry solids can be calculated from the relationship:

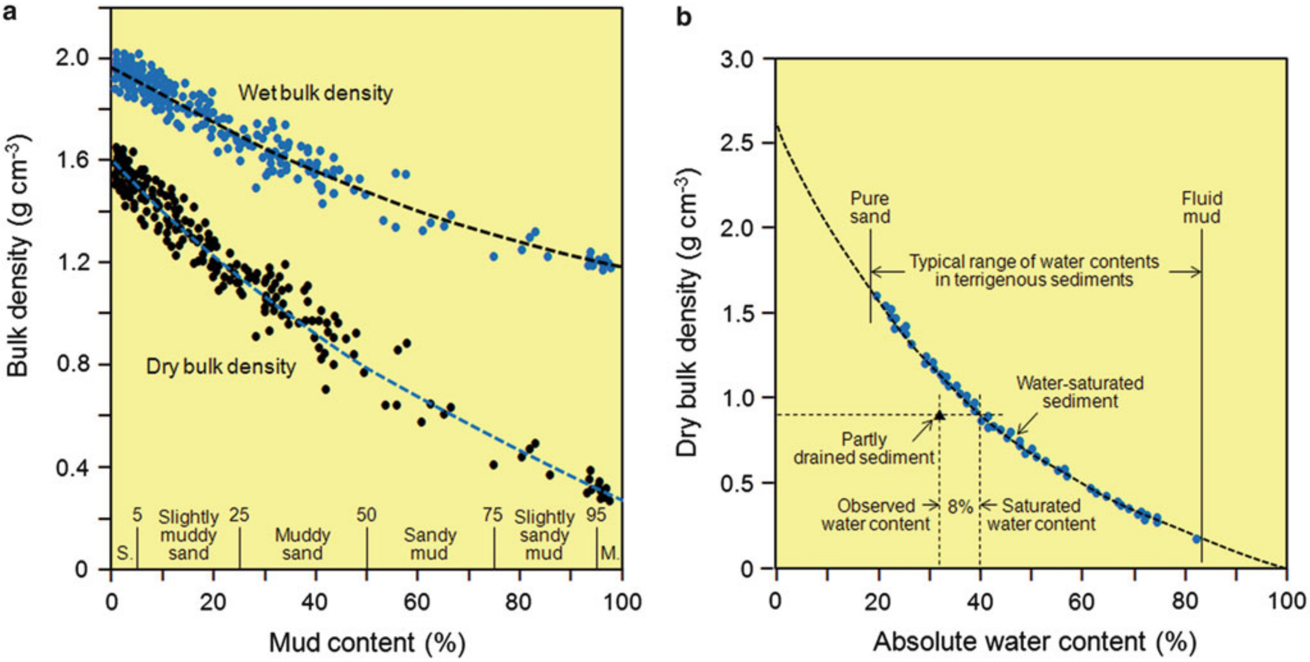
$$\rho_s = M_d/V_d \quad (5)$$

where M_d is the mass of the dry solids and V_d is the volume of the dry solids. This is a good cross-check by which unusual mineral compositions, e.g., exceptionally high heavy mineral content, can be detected.

The wet bulk density denotes the ratio between the total mass of a water-saturated sample and the volume of the water-saturated sediment, which is mathematically expressed by the relation:

$$BD_w = M_t/V_t \quad (6)$$

where BD_w is the wet bulk density (in g cm^{-3} or kg m^{-3}), M_t the total mass of the water-saturated sediment sample (g or kg), and V_t the volume of the water-saturated sediment.



Mass Physical Sediment Properties, Figure 3 (**a**) Wet and dry bulk density as a function of mud content in Wadden Sea sediments (Data from Flemming and Delafontaine (2000)). The sediment type classification follows the scheme of Flemming (2000). (**b**) Dry bulk density as a function of absolute water content for a sample suite ranging from pure sand to fluid mud. Note that this relationship has universal character for average terrigenous sediments composed of quartz, carbonate, and clay minerals. A departure from the universal trend exemplified by the *black triangle* can be explained by partial drainage due to either a lowering of the water table at the sampling site or the loss of water during sample handling.

The dry bulk density, by contrast, denotes the ratio between the mass of the dry solids and the volume of the water-saturated sediment sample. It is mathematically expressed by the relation:

$$BD_d = M_d/V_t \tag{7}$$

where BD_d is the dry bulk density (in g cm^{-3} or kg m^{-3}), M_d the mass of the dry solids (g or kg), and V_t as defined above. In saline environments, a correction for the salt content in the pore water has to be implemented (cf. section on "Water content" above). Because the values of BD_w and BD_d overlap (cf. Figure 3a), it is imperative to indicate which of the two one is dealing with. In practice it is more difficult to determine the wet bulk density because sediments are not always water saturated or because water may be lost during sampling, storage, or subsequent handling. Fortunately the two parameters are directly proportional, and it is therefore possible to calculate one from the other. For example, to determine the wet bulk density from the dry bulk density of intertidal Wadden Sea sediments, the following equation ($r = 0.9948$; $n = 337$) can be used (cf. Flemming and Delafontaine, 2000):

$$BD_w = 0.9903394 + 0.6211214\ BD_d \tag{8}$$

Due to the high correlation between bulk density and water content ($r = 0.9991$), both wet and dry bulk densities of average terrigenous sediments (grain density of 2.65 g cm^{-3}) can safely be determined indirectly from the absolute water content, the relations having universal character for typical terrigenous sediments (cf. Figure 3b). The advantage of this procedure is that it is independent of any local bulk density/mud content relationship. Thus, to calculate dry bulk densities from water contents, the following equation is recommended (Flemming and Delafontaine, 2000):

$$BD_d = 2.6596369 - 0.0886164\ W_a + 0.0088041\ W_a^{1.5} - 0.0002594\ W_a^2 \tag{9}$$

with notations as defined above. Similarly, wet bulk densities can be calculated from water contents by using the following equation ($r = 0.9902$) (modified after Delafontaine et al., 2004):

$$BD_w = 2.643046366 - 0.05862255\ W_a + 0.006559454\ W_a^{1.5} - 0.00023421\ W_a^2 \tag{10}$$

with notations as defined above.

The wet bulk density, however, can also be calculated on the basis of various assumptions concerning the physical nature of the sediment and the water. For this purpose the following equation can be used:

$$\mathrm{BD_w} = \rho_s(1-\varphi) + (\rho_w\varphi) \qquad (11)$$

where ρ_s is the density of the dry solids, ρ_w the density of the water, and φ the porosity. The procedure for calculating the density of the dry solids has already been introduced (Eq. 5). The porosity is calculated from Eq. (16) (see below).

Bulk density is an important parameter used for a variety of purposes, particularly in standing stock and flux studies where the individual contributions of various sediment components must be quantified. As will be shown below, it is in this context that the content versus concentration issue becomes particularly relevant.

Shear strength

The shear strength of sediment is a measure of the force required to overcome the internal friction (in sands) or cohesion between particles (in sand/silt/clay mixtures) and is commonly measured with an appropriately calibrated vane shear apparatus. In effect, the instrument measures the resistance of the sediment to the rotational force exerted by a calibrated, spring-supported vane inserted into the sediment. For practical purposes, the yield force (or yield pressure) can be related to other parameters such as current shear along the surface of a sediment body, vertical load forces exerted by objects rolling, falling, or resting on its surface, or the expulsion of rising liquids and gases. It can be expressed mathematically as

$$\tau = c + (\sigma - u)\tan\phi \qquad (12)$$

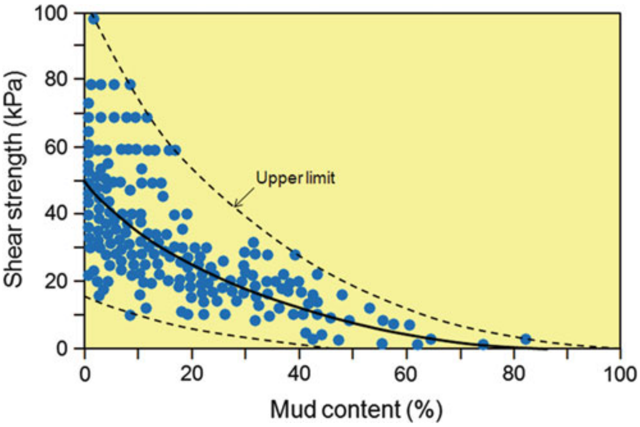
where τ is the shear strength, measured either as a pressure (e.g., in kPa) or as a force (e.g., in kg m^{-2}), and c is a measure of cohesion, σ the total stress, u the pore water pressure, and ϕ the angle of internal friction or shearing resistance.

An example of typical shear strengths in intertidal sediments along a sand–mud gradient of the Wadden Sea is presented in Figure 4 (cf. Flemming, 2012). Maximum and average shear strengths decrease markedly as the mud (and water) content increases. The largest variations, however, are recorded in sand. This trend is due to an increasing variability in local grain-packing densities as the sediment becomes sandier, low packing densities in fluidized sand, for example, having low shear strengths, whereas firmly packed sands, e.g., wave-compacted beach sand, exhibit high shear strengths.

Void ratio

The void ratio (e) is a dimensionless value expressing the relationship between the volume of the pore space and the volume of the dry solids:

$$e = V_v/V_d \qquad (13)$$



Mass Physical Sediment Properties, Figure 4 Shear strength as a function of mud content in intertidal Wadden Sea sediments (Data from Flemming (2011)). Note the progressive decrease in both mean values and overall range with increasing mud content. The diagram does not include measurements from compacted (i.e., partly dewatered) mud located above the mean high-water level, which can have shear strengths exceeding even those of compacted sand.

where V_v is the volume of the pore space and V_d is the volume of the dry solids in an undisturbed sediment sample. It can also be calculated from the following equations if, for example, the volume of the dry solids is unknown:

$$e = [(\rho_s\rho_w V_t)/M_d] - 1 \qquad (14)$$

or

$$e = \rho_s(W_r/100) \qquad (15)$$

with parameters as defined above.

Porosity

The porosity (φ) is the ratio between the pore volume (V_v) and the total volume (V_t) of an undisturbed sediment sample expressed as a percentage (%) and is defined as

$$\varphi = 100(V_v/V_t) \qquad (16)$$

with parameters as defined above. The pore space or volume (V_v) can be calculated from the relationship:

$$V_v = 100[e/(1+e)] \qquad (17)$$

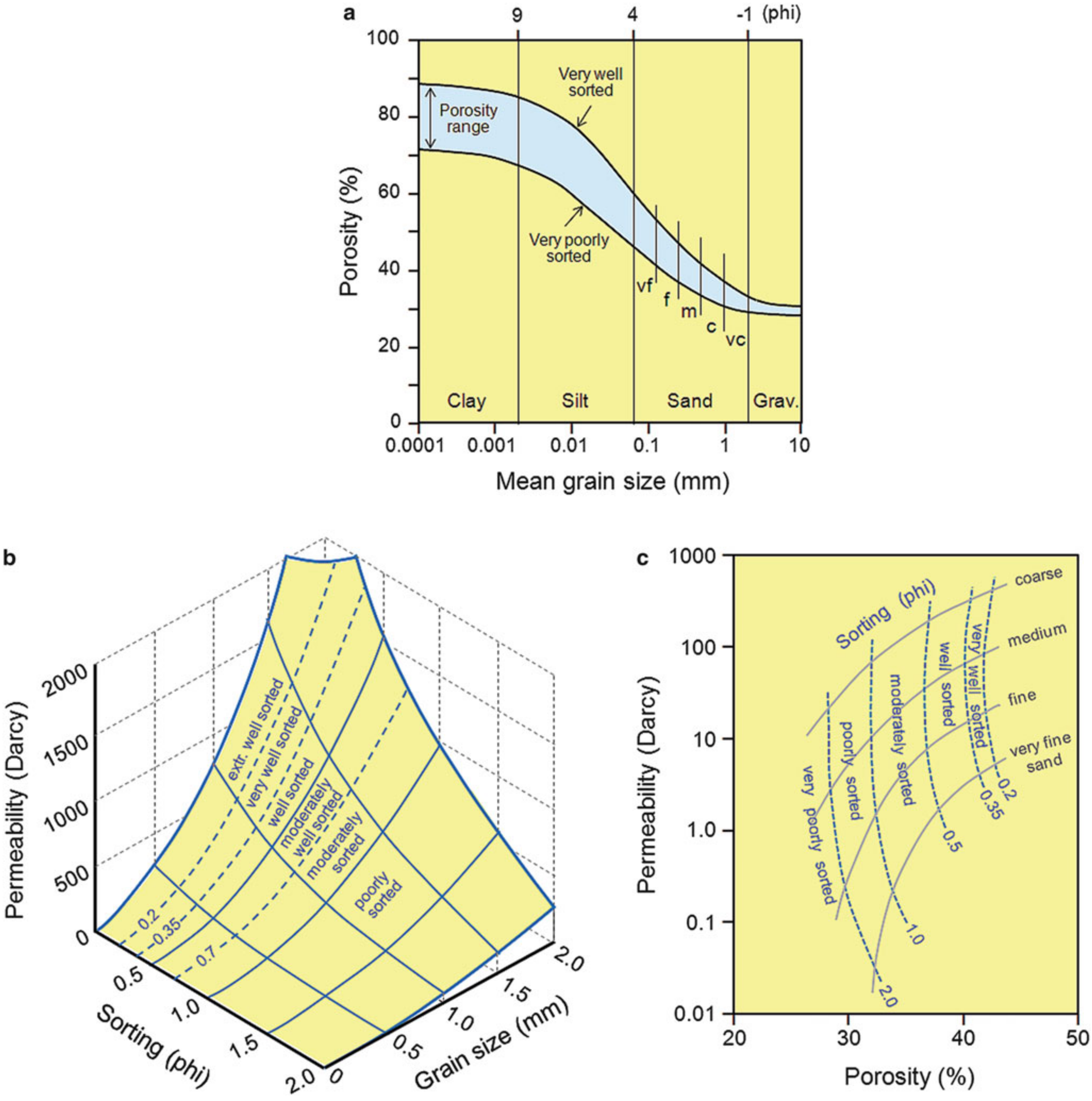
Porosities and void ratios can be converted from one into the other by the equations:

$$\varphi = e/(1+e) \qquad (18)$$

and

$$e = \varphi/(1-\varphi) \qquad (19)$$

Porosity is an important parameter in the study of, for example, sediment–organism relationships, sediment



Mass Physical Sediment Properties, Figure 5 (**a**) Porosity as a function of mean grain size. Note the increasing porosity with decreasing grain size, and the higher porosity in well sorted as opposed to poorly sorted sediment. (**b**) Permeability as a function of mean grain size and sorting (Modified after Krumbein and Monk (1942)). (**c**) Permeability as a function of porosity in relationship to mean grain size and sorting (Modified after Selley (1985)).

biogeochemistry, and material fluxes into and out of a sediment body. The porosity is a function of both grain size and sorting (standard deviation of a particle-size distribution), the porosity increasing with decreasing grain size (cf. Figure 5a). In addition, well-sorted sediment of a given mean grain size has a larger porosity than more poorly sorted counterparts because, in the latter case, finer particles increasingly occupy the pore space between larger particles as sorting decreases.

Permeability

The permeability is proportional to the rate at which water can flow through a given cross-section and distance or height of a porous or granular material. The quantity of fluid measured in this way is, in addition, directly proportional to the pressure difference at the two ends of the experimental setup and inversely proportional to the dynamic viscosity of the fluid. This relationship is expressed by the following equation:

$$Q = k(Cp/\eta L) \qquad (20)$$

where Q is the flow rate (in cm^3 s^{-1}), k is the proportionality factor representing the permeability and is measured in darcy (1 darcy being equivalent to 1 cm^3 of fluid having a viscosity of 1 cP flowing through a cross-section of 1 cm^2 at a pressure difference of 1 atm), C is the cross-section of the core tube (in cm^2) containing the sediment, p is the pressure difference at the two ends of the tube (in atm), η is the dynamic viscosity of the fluid (in centipoise), and L is the length of the tube (in cm). As k is the unknown variable in this equation and Q is determined for known values of C, p, η, and L, the equation can be rearranged to be solved for permeability:

$$k = Q/(Cp/\eta L) \qquad (20a)$$

As illustrated in Figure 5b, permeability is strongly affected by grain size and sorting of sediment. It rapidly increases with increasing grain size and improving sorting, eventually reaching several thousand darcy in well-sorted gravels. As the same grain-size parameters also control the porosity, it stands to reason that permeability should correlate with porosity as a function of grain size and sorting. This complex relationship is illustrated in Figure 5c.

Atterberg limits

The Atterberg limits (cf. Casagrande, 1948) are rheological parameters used in the classification of soils, being of particular importance in engineering geology and constructional engineering. They define the transitions from the liquid state to the plastic state of a soil (liquid limit, w_l), from the plastic state to the semisolid state (plastic limit, w_p), and from the semisolid state to the solid state (shrinkage limit, w_s). These transitions are a function of the relative water content of a soil (definition see above) which, in turn, is a function of soil composition and, especially in the presence of clay minerals, also their affinity to attract water. The limits can thus vary strongly from one soil to another, particularly where clay contents are high.

The liquid limit is determined by application of a standard mechanical device specifically designed for that purpose (cf. Lambe, 1951). It is defined by the water content and the number of calibrated hammer blows required to close a groove of specified depth and length in a soil test sample. The plastic limit is reached when thin threads of a soil 1/8 in. (~0.32 mm) in diameter begin to crumble when manually rolled, whereas the shrinkage limit is defined by the water content after just enough water has been added to fill all the voids of a dry soil sample. Typical Atterberg limits for selected clays, with sodium (Na) as the exchangeable ion (in view of estuarine salinity conditions), are listed in Table 1. The individual limits, especially the liquid limit (e.g., Faas, 1991), may vary strongly in the presence of other dominant exchangeable ions and the nature of the particle framework structure (or fabric) at the transition from fluid-supported to particle-supported hindered settling suspensions.

Mass Physical Sediment Properties, Table 1 Atterberg limits in terms of relative water contents for three clays with Na as the dominant exchangeable ion (From Lambe and Whitman, 1969)

Clay mineral	Liquid limit, w_l	Plastic limit, w_p	Plasticity index, I_p	Shrinkage limit, w_s
	(%)	(%)	(%)	(%)
Kaolinite	53	32	21	26.8
Illite	120	53	67	15.4
Montmorillonite	710	54	656	9.9

Table 1 also indicates that particular indices play important roles in the assessment of rheological soil behavior. The most important are the plasticity index (I_p), which expresses the difference between the liquid limit and the plastic limit ($I_p = w_l - w_p$), the so-called flow index (I_f) represented by the slope of the "flow" curve derived by plotting the water content against the number of blows imparted until the liquid limit is reached (see above), and the toughness index (I_t) represented by the ratio between the plasticity and flow indices ($I_t = I_p/I_f$).

The definitions of the rheological limits between the various soil states are quite arbitrary and hence difficult to interpret in physical terms. However, they remain useful empirical indicators for the expected behavior of a soil and are commonly applied in constructional engineering and engineering geology.

Carbonate content

The carbonate content of a sediment sample can be determined by a variety of methods, depending on the overall accuracy which is required. The simplest, albeit least accurate, method is to digest the carbonate fraction of a sediment sample of know weight by adding sufficient hydrochloric acid. After effervescence stops, the sample is rinsed in fresh water, dried, and reweighed; the weight loss is expressed as a percentage of the initial mass corresponding to the carbonate content. This procedure is quite adequate to, for example, generate $CaCO_3$ distribution maps of environments relatively rich (>5 %) in bioclastic material. Somewhat greater accuracy and better reproducibility is achieved by application of the so-called carbonate bomb (Müller and Gastner, 1971), which makes use of a defined aliquot of finely ground sample placed into a small chamber containing a separate container filled with hydrochloric acid. After sealing the chamber, the acid is released for digestion of the carbonate. The pressure of the CO_2 gas generated by the digestion process is indicated by a calibrated pressure gauge attached to the chamber. It is proportional to the carbonate content of the sample at a standard temperature and atmospheric pressure, the reading having to be corrected for changing ambient temperatures and atmospheric pressures.

In cases where greater accuracy and precision is required, the carbonate content should be determined by means of industrial CHN analyzers which enable the measurement of total carbon (C_{tot}) and organic carbon (C_{org}). The carbonate content (expressed as wt%) is then calculated by the stoichiometric equation:

$$CaCO_3 = 8.33\left(C_{tot} - C_{org}\right) \tag{21}$$

The application of a CHN analyzer has the additional advantage of providing high-precision measurements of the particulate organic carbon (POC) and nitrogen (PON) contents of the sediment.

Selected applications and potential pitfalls

Sampling and monitoring strategies

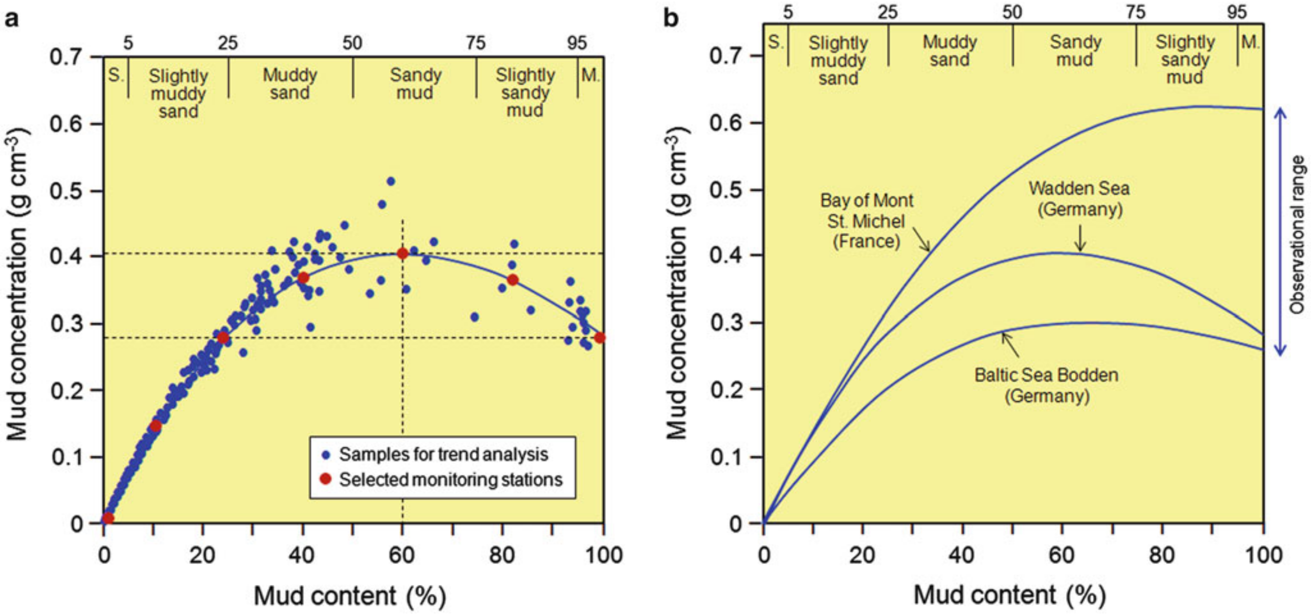
The study of heterogeneous environments, such as estuaries, requires having a minimum number of samples which are representative of the environment as a whole (e.g., Underwood and Chapman, 2013). The ultimate aim of defining a minimum sample number is to minimize effort and cost without sacrificing or violating scientific standards. This requires carefully designed sampling strategies based on rational criteria (Flemming, 2011). Clearly, different disciplines have different constraints regarding the effort and cost issue. Sedimentologists, for example, are least constrained regarding sample numbers because sampling, processing, and analysis are relatively rapid and cost-efficient. Geochemists and biogeochemists, by contrast, are far more constrained in this respect, while biologists and microbiologists are probably the most constrained. In view of this, initial pilot studies focusing on mass physical sediment properties can provide a rational decision basis for almost any situation, besides being particularly cost-effective.

For this purpose the use of specially prepared short coring vials of known volume is recommended. Samples should be taken along transects of sand–mud gradients (preferentially from 0 to 100 % mud content) in such a way as to minimize sediment disturbance, especially avoiding loss of sediment and pore water during both sampling and subsequent handling. The geographic locations of sampling sites should be determined by GPS. Immediately after sampling, the full vials are cleaned and dried, and the drainage holes sealed by tape before being stored in a coolbox. The ultimate aim is to have replicate samples at intervals of at least 5 % mud content, which would amount to a minimum of 42 samples along a 0–100 % mud content gradient. Excessively large gaps between consecutive mud content values identified after the initial survey may have to be filled in subsequent sampling campaigns. In the laboratory the (clean and dry) sealed vials are weighed before further processing (the mass of the empty vial, including the clamp lid and tape, having later to be subtracted from the total mass in order to calculate both the wet and dry mass of the samples). Mass physical sediment properties (wet and dry bulk density, water content, mud content) are then determined on the whole sample using standard procedures and the relationships outlined above. If desired, other parameters such as organic matter or POC and $CaCO_3$ contents can subsequently be determined on appropriate subsamples.

The outcome of such a sampling campaign is illustrated in Figure 6a for the case of the Wadden Sea (cf. also Flemming, 2011). The diagram shows the dry mass concentration of mud (g cm^{-3}) as a function of mud content (dry weight %), the regression curve representing the average trend of the survey area (cf. also Figure 1). It should be noted that any component linked to the mud fraction (e.g., organic matter or POC, heavy metals, pollutants) will show a similar trend in its concentration. On this basis a rational decision concerning the selection of a minimal number of representative locations for more detailed investigations can be made. In the case of Figure 6a, for example, a well-founded scientific strategy would be to select the seven study sites represented by the red dots along the regression curve. This rationalized procedure optimizes time and effort without jeopardizing scientific objectives and standards.

Selected applications

As pointed out above, bulk densities of similar sediments may vary between environments for a variety of reasons. As a consequence, average mass concentration curves may differ greatly from the Wadden Sea example used here to illustrate the basic principle. In order to assess the potential range of such variability, the Wadden Sea mass concentration curve has been contrasted with those from two other, very different estuarine environments, namely, the Bay of Mont St. Michel (Gulf of St. Malo, France) and a backwater lagoon (Greifswalder Bodden) from the Baltic Sea coast of Germany (Figure 6b). The two alternate curves, which can be taken as representing approximate upper and lower range limits, differ from the Wadden Sea curve in that the Bay of Mont St. Michel curve shows no and the Bodden curve only a faint mud concentration peak at intermediate mud contents. Depending on local bulk densities, individual curves may thus vary from strongly peaked to plateau-like shapes. It is for this reason that the calibration curve of one environment (e.g., the Wadden Sea) cannot simply be transposed to another environment but that separate calibrations are needed in each case.



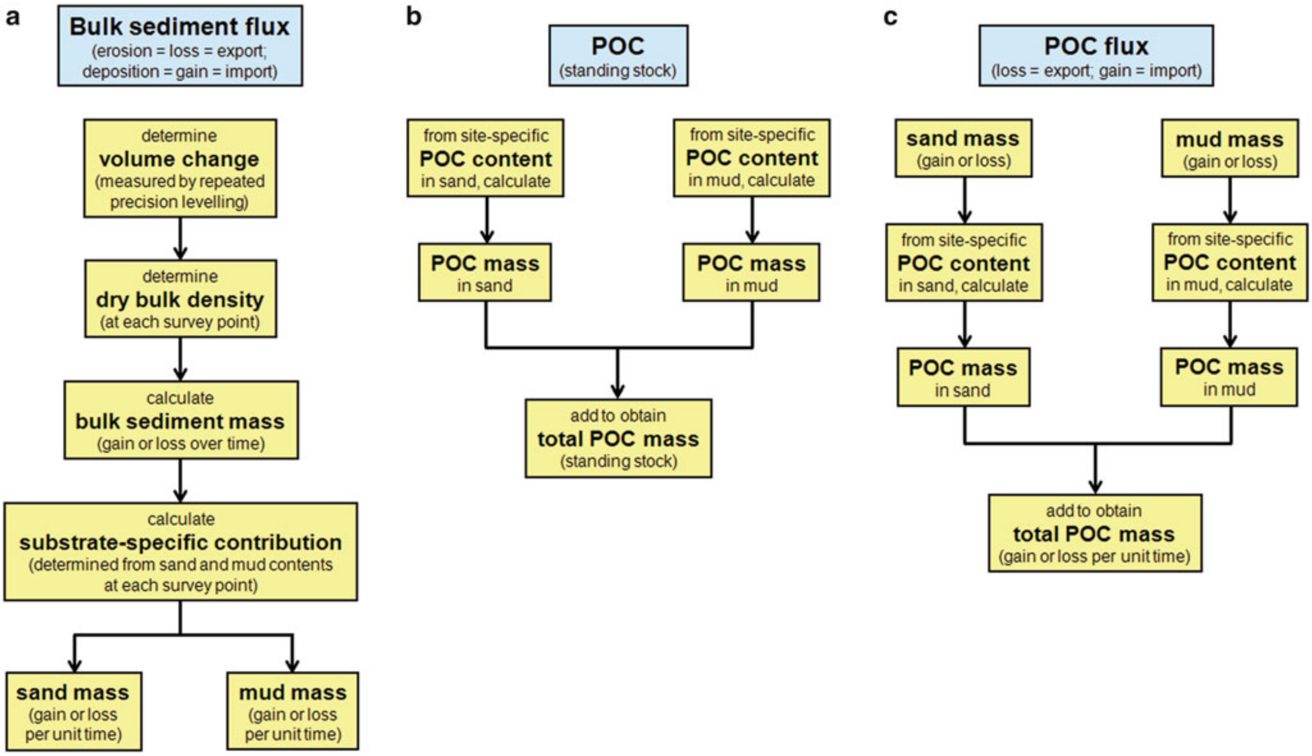
Mass Physical Sediment Properties, Figure 6 (**a**) Typical trend of the mud mass concentration as a function of mud content in intertidal Wadden Sea sediments (Data from Flemming and Delafontaine, 2000; cf. also Flemming, 2011). Note the reversal of the trend after the average mud concentration (4.1 g cm^{-3}) reaches a peak at a mud content of 60 %; at 100 % mud content the mud concentration is only about 2.8 g cm^{-3}, i.e., identical to that at a mud content of about 23 %. (**b**) Examples of reversing or plateauing mud mass concentration trends observed along mud content gradients in three estuarine environments, defining the currently known range covering especially low (Bodden backwaters) to high values (Bay of Mont St Michel tidal flat). The sediment type classification follows the scheme of Flemming (2000).

Once established, calibration curves can be used, for example, to calculate sediment fluxes over time from volume changes determined for specific areas within estuaries, either by repeated multibeam bathymetric surveys (in subtidal settings) or precision leveling (in intertidal settings). The procedure for converting volumetric changes into sediment masses, either positive (gain) or negative (loss), is illustrated in the flow chart of Figure 7a (cf. Bartholomä et al., 2000). Besides the total sediment flux, the flux of the sand and mud fractions can also be determined if the contributions of these to the total sediment are known. This also applies to any other known sediment component, as illustrated for standing stocks in Figure 7b and fluxes of POC in Figure 7c (cf. Delafontaine et al., 2000). The same general procedures can be applied to other organic or inorganic substances, provided their contents are known.

Potential pitfalls

As pointed out above, contents (masses per unit mass) and concentrations (masses per unit volume) are often confused or even understood to mean the same thing. This can lead to serious errors where parameters of the one, e.g., species numbers per square meter (a measure of concentration), are directly related to parameters of the other, e.g., percent organic matter (a measure of content), over a range of sand–mud mixtures. By doing this, it is overlooked that the dimensional incompatibility will cause misrepresentations and faulty assessments.

To highlight the nature of this problem, a mud concentration model is contrasted with a mud content model for a sand–mud gradient in the Wadden Sea (Figure 8). It can be viewed as representing either a transect or a vertical sediment column (core) ranging in composition from pure sand (yellow boxes at bottom) to pure mud (dark blue boxes at top). Due to the progressive decrease in bulk density with increasing mud content, the sediment mass in a unit volume of sediment (e.g., 1 l or 1 dm^{3}) decreases from about 1.5 g cm^{-3} in sand to about 0.3 g cm^{-3} in mud (mud concentration model). Any other sediment component (e.g., organic matter, POC, heavy metals, toxic substances) would decrease proportionally and, in this example, follow the trend defined by the blue line representing the mud concentration (graph on the right). Clearly, when relating parameters such as species numbers per m^{2} to the potentially available food, the concentration of organic matter or POC has to be chosen for meaningful comparison. If organic matter or POC contents were used instead, as is commonly observed in the literature, one would be comparing the species

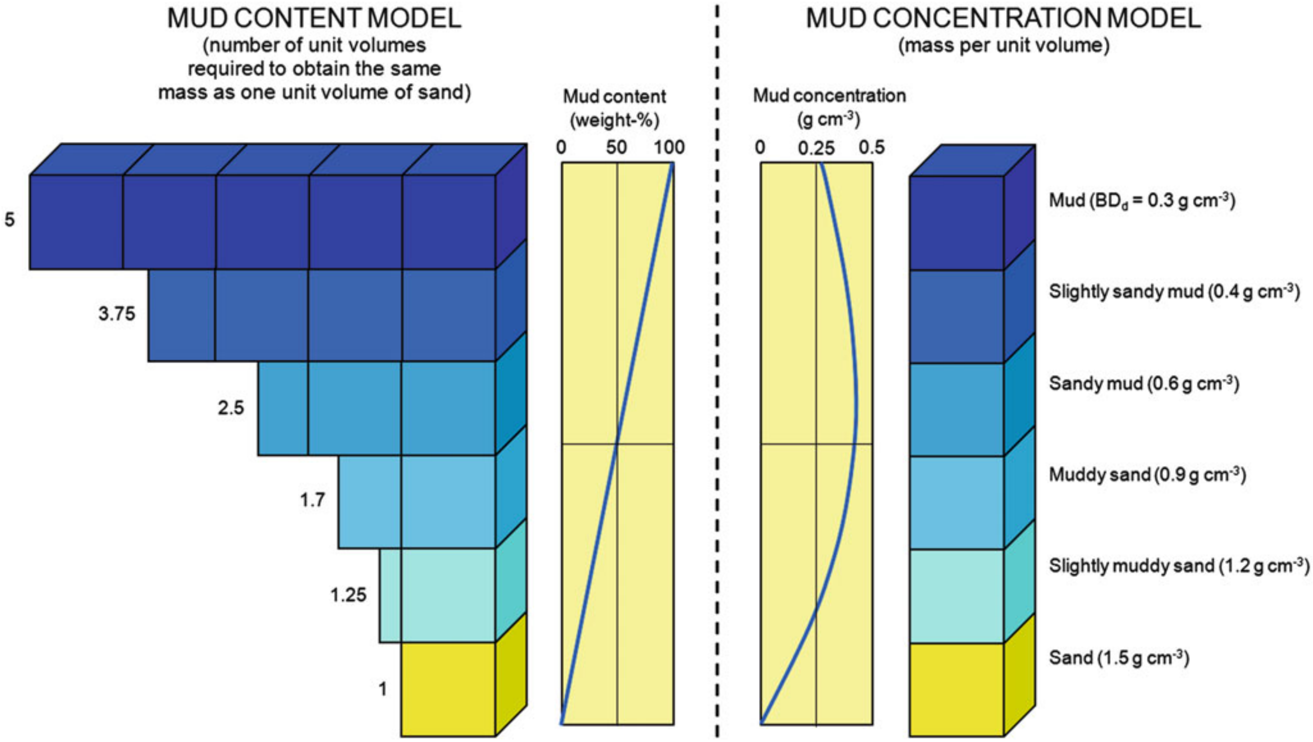


Mass Physical Sediment Properties, Figure 7 (**a**) Schematized procedure for the calculation of bulk sediment fluxes in terms of import to or export from a defined area. Note the role of bulk density in converting volume changes into corresponding masses of bulk sediment as well as specific sediment fractions. This would also apply to any other sediment component for which the contents are known (Based on Bartholomä et al., 2000). (**b**) Calculation of standing stocks of particulate organic carbon (POC). (**c**) Calculation of POC fluxes. Note that the calculations continue from the site-specific losses or gains in sediment masses determined by the procedure illustrated in (**a**) (Based on Delafontaine et al., 2000).

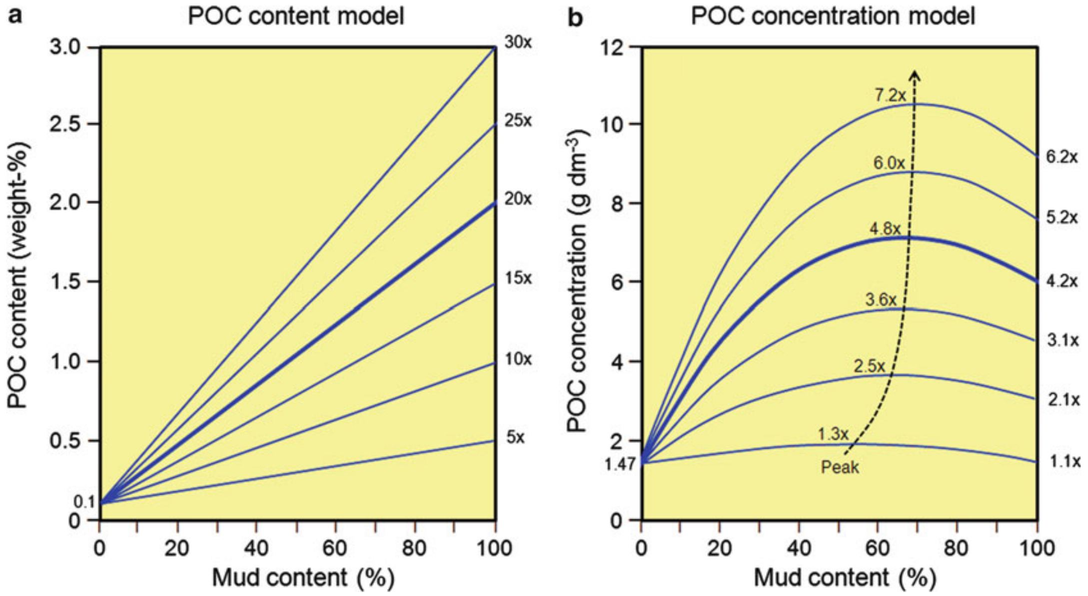
numbers in 1 m^2 at all sites with the amount of organic matter or POC in increasingly larger areas (or volumes), as illustrated in the mud content model on the left side of Figure 8. In effect, the amount of potentially available food would be greatly overestimated, especially at high mud contents.

This pitfall becomes even clearer when comparing the POC content model of Figure 9a with the corresponding POC concentration model in Figure 9b, where the average trends in content and concentration of a variety of sediment POC loadings are shown. A globally common trend in POC contents is represented by the thick blue line in Figure 9a, with about 0.1 % POC in sand and 2 % in mud. Applying the Wadden Sea mud concentration model, the corresponding trend reaches a peak at a mud content of about 68 % (Figure 9b), a factor of 4.8 larger than for pure sand. At 100 % mud content, by contrast, the POC concentration has decreased to a factor of only 4.2 relative to pure sand. Clearly, the factors in the POC concentration model stand in stark contrast to those in the POC content model, the highest amount of potential food being available in mixed sediment, not in pure mud as suggested by the POC content model. This has far-reaching implications for the assessment of, among others, animal–sediment relationships in terms of species distribution models, standing stocks and turnover of available food or higher-quality components thereof, and ecotoxicological impacts.

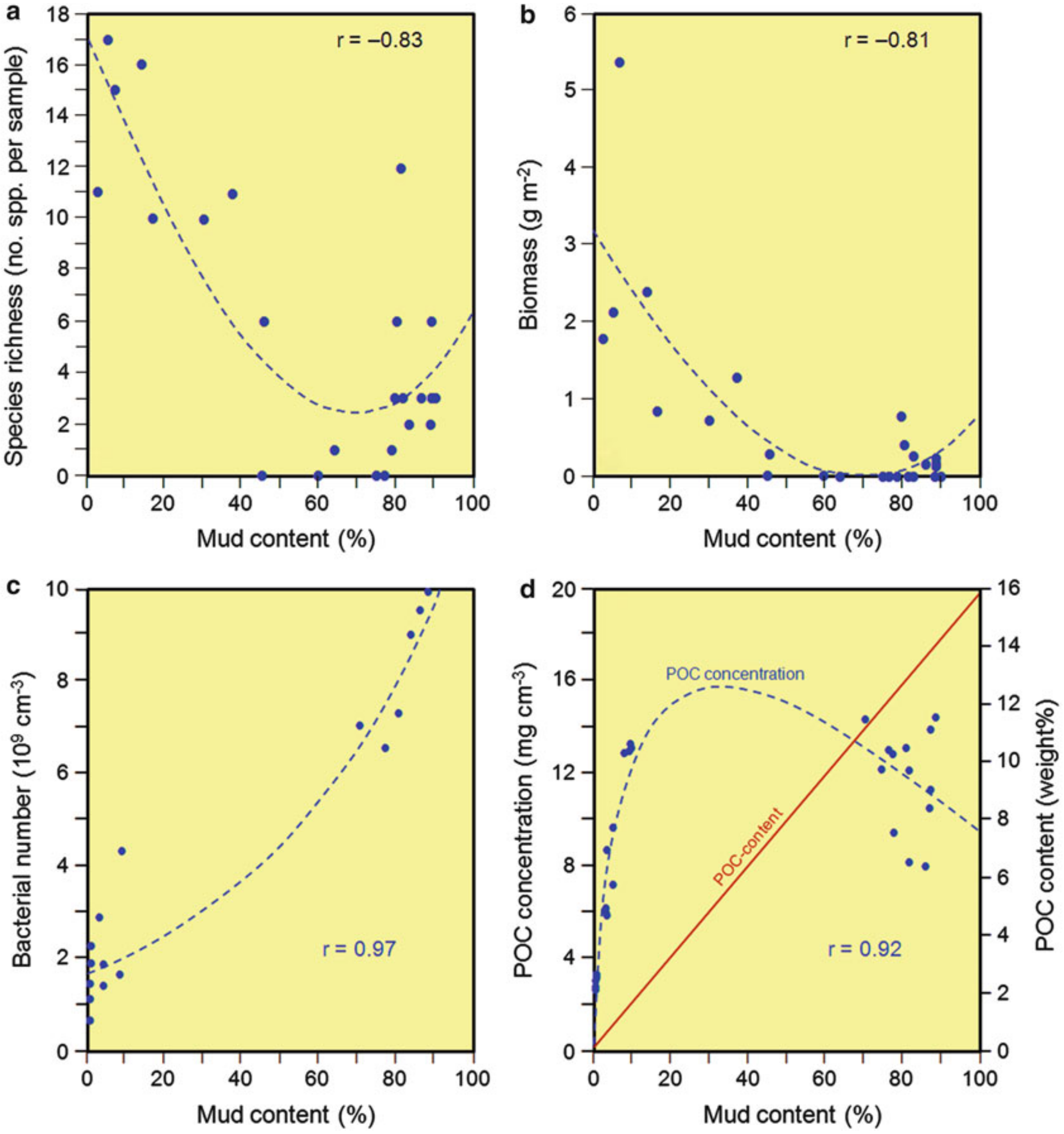
Ecotoxicological effects are a case in point. Studying the effect of sediment contamination by toxic substances on benthic animals in dead-end canals of coastal lagoons along the Delaware and Maryland coasts of the USA, Maxted et al. (1997) documented a marked overall decrease in species richness and biomass with increasing mud content. Assuming highest contamination to be coincident with highest mud contents, the applied linear regression model yielded statistically significant correlation coefficients of $r = -0.75$ and $r = -0.73$, respectively. In so doing, these authors evidently did not recognize the counterintuitive signs of ecological recovery visible at mud contents exceeding about 80 %. Indeed, when applying a curvilinear regression model (Figure 10a, b), the present authors achieved considerably better correlations ($r = -0.83$ and $r = -0.81$, respectively), demonstrating that a concentration model of the type illustrated in Figure 9b, which accounts for the partial reversal of the toxic effect at high mud contents, would have been more meaningful.



Mass Physical Sediment Properties, Figure 8 Mud content model (*left*) contrasted with mud concentration model (*right*) as a function of sediment type. The models can be viewed as representing an area or a vertical sediment column. In the case of the content model on the *left*, note the progressive increase in sediment volumes (numbers on *left*) as a function of fining sediment type required to obtain the same mass as that of the unit volume of sand (*yellow box* at bottom). This volume increase is explained in the concentration model on the *right-hand* side by the changing bulk densities of the sand–mud mixtures (BD_d denotes dry bulk density).



Mass Physical Sediment Properties, Figure 9 (**a**) Content model showing average trends of POC content as a function of mud content for a number of increasing POC loadings. (**b**) Concentration model showing the corresponding loadings assessed in terms of POC concentrations. Note the vastly different factors at 100 % mud content in the two models and the peak concentration values between about 53 % and 70 % mud content in the concentration model.



Mass Physical Sediment Properties, Figure 10 (**a**) Species richness and (**b**) biomass as a function of mud content observed by Maxted et al. (1997) in estuarine sediment contaminated with toxic substances (Delaware and Maryland coastal bays). Improved correlation coefficients (with r = −0.83 instead of r = −0.75 and r = −0.81 instead of r = −0.73, respectively) were achieved by the present authors when applying nonlinear regression instead of linear regression. This demonstrates that a concentration model of the type illustrated in Figure 9b, which reveals the reversal of the toxic effect at high mud contents, would have been more appropriate. (**c**) Bacterial numbers and (**d**) POC concentration as a function of mud content in Bodden sediments, Baltic Sea (Modified after Köster et al. (2005)). Note the lack of correlation between the two parameters, which is in stark contrast to the high (but spurious) correlation between microbial biomass and POC content (*red* regression line and scale on right) observed in an earlier study (Köster et al., 1997).

Bird and Duarte (1989) warned about generating spurious trends devoid of scientific meaning when studying relationships between bacteria measured as numbers or masses per unit area or volume (concentrations) and organic matter measured as masses per unit mass (contents) of sediment. An example is illustrated in Figure 10c and d (based on data from Köster et al., 1997, 2005). Köster et al. (1997) compared bacterial number concentrations (determined per unit sediment volume) with POC contents (determined per unit sediment mass), the high correlation being interpreted as evidence for carbon limitation of microbial biomass development. Recognizing the inherently spurious nature of that relationship, Köster et al. (2005) subsequently compared bacterial number concentrations (Figure 10a) with POC concentrations (determined per unit sediment volume; Figure 10d).

The obvious lack of correlation was now interpreted as evidence to the contrary, i.e., that particulate organic carbon was not the anticipated limiting factor in microbial growth and activity but that these were more likely controlled by the availability of dissolved organic matter and inorganic nutrients.

Summary

Unfortunately, dimensional misrepresentations of the type outlined in this contribution pervade the scientific literature (cf. Flemming and Delafontaine, 2000), such errors having found their way into models (e.g., Paarlberg et al., 2005; Borsje et al., 2008), manuals (e.g., Gray and Elliot, 2009), and environmental guidelines and directives (e.g., Bjørgesæter and Gray, 2008). On the other hand, since the content versus concentration issue was raised by Flemming and Delafontaine (2000), an increasing number of investigations and textbooks have recognized the need to switch to volumetric units for intercomparisons of sedimentary abiotic and biotic parameters (e.g., Perkins et al., 2003; Köster et al., 2005; Tolhurst et al., 2005; Jesus et al., 2006; Chapman and Tolhurst, 2007; Tolhurst et al., 2008; Giere, 2009; Leipe et al., 2011; Adams et al., 2012; Kenny and Sotheran, 2013). This has led to sometimes surprising interpretations and questions which would otherwise have remained obscure. Meeting such challenges will be greatly facilitated by recent progress in the development of remote sensing and other instrumentation for the in situ appraisal of mass physical sediment properties at various spatial scales (e.g., de Groot et al., 2009; Jacobs et al., 2009; Ha et al., 2010; Barry et al., 2012; L'Esperance et al., 2013).

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Cross-references

Benthic Ecology
Estuarine Sediment Composition
Infauna
Marsh Sediment Toxicity
Nonpoint Source Pollution
Sandflat
Sediment Budgets
Sediment Compaction
Sediment Grain Size
Species Richness
Species Zonation
Subaqueous Soils

MEAN SEA LEVEL

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Synonyms

Average sea surface height

Definition

Mean sea level (MSL) is the datum representing the average height of the surface of the sea for all stages of the tide, typically computed by local hourly water level heights over a 19-year period.

Description

The mean sea level is used as a reference for dry-land elevations. It varies slightly from one place to another on earth because the sea surface conforms to the earth's gravitational field, which varies from place to place. Using satellite altimeters, mean sea level changes across the globe are measured and averaged to detect trends in the global mean sea level as an indicator of climate change. Mean sea level changes over short and long time periods. Tides, winds, and seasonal variations are examples of causes of short-term changes. Climate change, uplift, and subsidence of the earth's surface are examples of causes of long-term changes. For example, 20,000 years ago during the last ice age, the mean sea level was ~120 m below current levels due to the retention of water in ice sheets that covered parts of North America, northern Europe, and other regions on earth.

Cross-references

Climate Change

MEIOFAUNA

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Definition

Meiofauna are defined by Guiere (2009) as small mobile and sometimes sessile benthic organisms ranging in size between that of the microfauna and macrofauna. They include benthic organisms that pass through a sieve mesh size of 500 μm (or 1,000 μm) as the upper limit, and are retained on a sieve mesh size of 44 μm (or 63 μm) as the lower limit. Meiofauna are often found moving between sediment interstices, but also occur on hard surfaces, algae, coral rubble, or other surfaces. There are both temporary and permanent members of meiofaunal communities in estuarine environments. Although the meiofauna are diminuitive forms, they can play a role in the bioturbation of bottom sediments. Important groups of meiofauna found in estuaries include the rotifers, gastrotrichs, nematodes, polychaetes, tartigrades, copepods, turbellarians, and ostracods.

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Cross-references

Benthic Ecology
Infauna
Macrofauna
Microfauna
Soft Sediment Communities

MICROBIAL DEGRADATION

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Synonyms

Biodegradation; Mineralization

Definition

Microbial degradation refers to microbial action that drives biogeochemical cycles, notably with respect to major elements (carbon, nitrogen, sulfur, phosphorus, iron), as well as the breakdown of contaminants that can be used as a microbial food source or substrate.

Description

In marine ecosystems, dissolved organic matter (DOM) constitutes the largest amount of biologically reactive organic carbon equal to the amount of carbon dioxide (CO2) in the earth's atmosphere and other sources (terrestrial or marine). Cycling of dissolved organic carbon significantly influences the global carbon cycle (Kirchman et al., 2009). A large amount of DOC derives from phytoplankton primary production (Khodse and Bhosle, 2011).

Biodegradation or microbial mineralization is an important process for the removal of riverine DOC in estuarine and coastal marine waters, where the biologically reactive fraction of the riverine organic matter may be partially or completely mineralized. Thus, many estuaries may be considered net heterotrophic systems, with respiration exceeding primary production (He et al., 2010). In many estuaries worldwide, a net loss of riverine particulate organic carbon (POC) has been reported, while the DOC often exhibits an apparent conservative behavior (Servais and Garnier, 2006).

In estuaries, biodegradation by heterotrophic bacteria can change rapidly along the estuarine gradient, showing a selective removal of carbohydrates and amino acids within the DOC pool in the upper reach and mixing zone and an autotrophic source of particulate carbohydrates (PCHO) in the lower estuary. These differences provide insight into the DOC process in estuaries (He et al., 2010).

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Cross-references

Bioremediation
Carbon Sequestration
Microbial Survivability

MICROBIAL SURVIVABILITY

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Synonyms

Adaptability to new habitats; Microbial viability, Potentially survivable marine organisms

Definition

Ballast water. Seawater loaded onto ships as ballast to maintain hull stability and propulsion efficiency.

Cyst. Highly survived state of phytoplankton from which plankton cells are reproduced by germination.

Ten of the most unwanted marine organisms. Ten kinds of marine organisms listed by the International Maritime Organization as highly survivable, ecological threats.

Introduction

The first iron steamship, *Aaron Manby*, was built in the nineteenth century. Thereafter, seawater was commonly used as ballast on ships. Massive transport of seawater is still continued by ships, and many marine organisms are transported across oceanic boundaries in the ballast water that is loaded on board. Modern developments in containerizing cargo, the increasing size of hulls, and high-performing seawater pumps also accelerated the utilization of seawater as ballast.

The International Maritime Organization (IMO) adopted the International Convention for the Control and Management of Ships' Ballast Water and Sediments in 2004 to determine the minimum viable cell numbers of pathogenic *Vibrio cholerae*, *Escherichia coli*, and intestinal *Enterococci* as standards for discharged ballast water (IMO, 2004). The IMO also listed ten of the most unwanted kinds of marine organisms. These organisms are *V. cholerae*, cladoceran water flea, mitten crab, toxic algae (red/brown/green tides), round goby, European green crab, Asian kelp, zebra mussel, North Pacific seastar, and North American comb jelly. Zebra mussel is one of the most successfully established invasive species in the Great Lakes, propagating inside seawater intake pipes and sometimes interrupting seawater uptake of power plants.

Potentially survivable marine organisms transported across oceanic boundaries affect the ecosystem of estuaries throughout voyages.

Effectiveness of ballast water exchange on the high seas for the prevention of nonnative microorganism invasion into estuaries

Dissolved oxygen and nutrients in seawater stored in a ballast tank are consumed by surviving marine organisms contained in the water during a voyage. Zooplankton cells have difficulty surviving in such conditions. Cysts of phytoplankton, however, can survive a voyage despite the low oxygen and poor nutrients in the ballast water stored in a dark ballast tank. In order to prevent dispersal of cysts across ocean boundaries via ships' ballast water, the IMO recommends ballast water exchange on the high seas before arrival at a foreign port.

Japan imports huge amounts of natural resources, such as crude oil, liquefied natural gas, iron ore, wood chips, etc., that arrive on mercantile ships from many countries. Therefore, Japan is one of the major ballast water export countries in the world because ballast water is discharged at the ports of call where the cargoes are loaded. The effectiveness of ballast water exchange in reducing plankton populations was examined using a liquefied natural gas carrier (110,000 gross tons) on the way to Qatar from Japan. The resulting reduction in the total number of phytoplankton cells by the exchange of ballast water on the high seas was confirmed by measuring the surface color of the ocean with a satellite remote sensor as well as direct counting of phytoplankton in samples taken on the ship (Kozai et al., 2006). In addition, the marine bacterial population can change markedly by replacing ballast water on the high seas. This process has been confirmed based on denaturing gradient gel electrophoresis analysis of samples taken on a coal carrier (58,098 gross tons) bound for Australia from Japan (Tomaru et al., 2010).

Surviving bacterial populations in sediments taken into a ballast tank are 10–100 times higher than those in ballast water (Mimura et al., 2005). Therefore, suspended sediments in ballast water should be controlled prior to the discharge of ballast water on the high seas or at ports of call. Development of onboard ballast water treatment systems is in progress (Wright et al., 2010; Nanayakkara et al., 2011; Abe and Mimura, 2013).

Using toxic dinoflagellates as a model indicator of microorganisms to trace biological invasion pathways

Marine organisms must survive several steps to become established in a new environment: (1) passing through a seawater pump when ballast water is loaded onto a ship, (2) remaining in a ballast tank during the voyage, (3) passing through the seawater pump when discharged with the ballast water, (4) adapting to a new environment, (5) reproducing and establishing a population, and (6) creating damage to populations of native species and the local economy. It has been pointed out that the blooming of toxic dinoflagellates is an intrinsic threat to coastal areas. An incidence of paralytic shellfish poisoning, which is closely related to the consumption of shellfish products that had accumulated alkaloid toxins, caused human illness.

These types of occurrences can cause enormous economic impact, and the ecological balance is affected by the blooming of the dinoflagellates.

Unlike the toxic *Alexandrium*, resting *Gymnodinium catenatum* has relatively resistant cell walls, making it possible to examine fossil records in sediments. The analysis of ^{210}Pb-dated samples taken from Australian coastal areas showed that the introduction of *G. catenatum* into southern Tasmanian waters occurred around 1972 (Hallegraeff, 1998). Populations of *G. catenatum* have been found in Australian ports since the 1990s. The appearance of *G. catenatum* coincides with the start of bulk woodchip exports from southern Tasmania to Japan.

The chemical structures of paralytic shellfish toxins produced by geographically different populations of *G. catenatum* are distinguishable from each other (Oshima et al., 1993). Their structures are not affected by temperature, salinity, or nitrate and phosphate concentrations in the culture medium. The major toxins, which have the *N*-sulfocarbamoyl group as a substituent and were produced by the species isolated in 1986 and 1988 from Tasmania, Japan, and the Galicia region of Spain, were C1 to C4 toxins as well as gonyautoxins 5 and 6. Only the Australian population produced the novel 13-deoxydecarbamoyl toxins.

Accumulated mutations in rDNA are known to reflect the prolonged geographic isolation and independent evolution within the region. Genetic analyses of the toxic dinoflagellates *Alexandrium tamarense* and *A. catenella* were made based on rDNA sequences and were applied to the interpretation of the biological invasion pathway. The small and large subunit rDNA sequences show the genetic diversity of *A. tamarense* and *A. catenella* living in Japanese coastal areas (Scholin et al., 1995). One possible thought is that the Eastern Asian species was contaminated with the Western European species prior to the 1970s, and diverse populations taken at ports of call in the Asian region were introduced into Australian waters via unloaded ballast water around the 1970s.

Survivability of pathogenic *Vibrio cholerae* in environments

Cholera is an epidemiologic syndrome caused by *V. cholerae*. The pathogenesis is classified into two groups based on the serogroups of O1 and O139 (Kaper et al., 1995). Furthermore, *V. cholerae* O1 is classified into two biotypes, classical and El Tor. The seventh pandemic of cholera, caused by *V. cholerae* O1 of the El Tor biotype, has been experienced since 1961 in locations around the world.

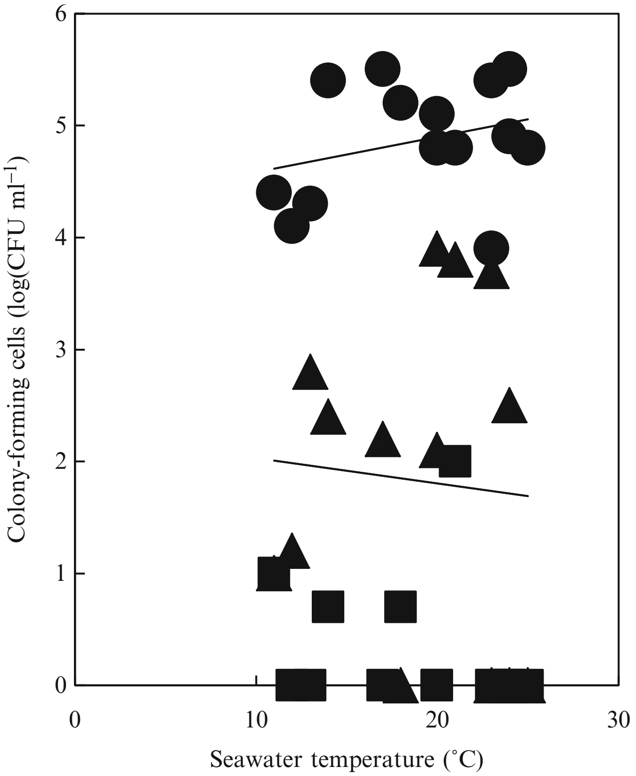
V. cholerae can exist in metabolically inactive states during which the cultivability of dormant cells declines but remains alive. The cells in such a "viable but nonculturable" state are thought to enhance survivability under harsh conditions. As for the process of becoming active following dormancy, a recent study on the germination of *Bacillus subtilis* spores suggests that cell wall muropeptides released from growing cells serve as the signal molecules for the initiation of growth (Dworkin and Shah, 2010).

V. cholerae of O1 and O139 serogroups were detected in more than 93 % of ships ($n = 15$ ships) calling at ports in Chesapeake Bay, USA (Ruiz et al., 2000). The numbers of *V. cholerae* cells having the O1 and O139 antigens in seawater samples were orders of 10^5 and 10^4 colony-forming units (CFU) liter^{-1}, respectively. Although their percentages were approximately 1 % compared with the total number of cells detected in the seawater ($n = 7$ ships), both serotypes had adhered onto plankton samples. The results indicate that pathogenic *V. cholerae* have continuously dispersed into coastal areas with discharged ballast water, increasing the risk of human infection with cholera.

Estimation of survivability of *V. cholerae* in Tokyo Bay following discharge with ballast water

More than 5,900 overseas vessels have arrived in Tokyo Bay in recent years. It is natural to assume that large amounts of ballast water containing pathogenic *V. cholerae* have been discharged in the area. Mimura and Miwa (2013) counted the total population of colony-forming marine vibrios in samples taken in Tokyo Bay. The numbers of non-sucrose-metabolized and sucrose-metabolized vibrio populations on thiosulfate-citrate-bile salt-sucrose agar plates were less than $10^{4.0}$ and $10^{2.0}$ CFU ml^{-1}, respectively, for all of the samples ($n = 6$) (Figure 1). No population that could make colonies at 37 °C was obtained on the plates, and the population of sucrose-metabolized marine vibrios was less than 0.16 % of the total bacterial populations for all of the samples taken in Tokyo Bay.

The estimation of survivability of *V. cholerae* after dispersal into the coastal area in Tokyo Bay was carried out using *Vibrio* sp. as a substitute strain of the pathogenic *V. cholerae* (Mimura and Miwa, 2013). The isolate used in the experiment has physiological and biochemical similarities with those of the pathogen, i.e., they can grow in 37 °C and metabolize sucrose as a carbon source (Mimura et al., 2005). Seawater samples were taken from Tokyo Bay from June 2009 to March 2010. When the cells on orders of 10^3 and 10^4 CFU ml^{-1} were added to raw seawater samples that had different numbers of native populations of $10^{3.5}$–$10^{6.1}$ CFU ml^{-1} ($n = 6$ seawater samples), it was confirmed that no *Vibrio* sp. cells survived 2 weeks of incubation at given temperatures, except in one instance. In the sample in which *Vibrio* sp. cells survived, the number of surviving $10^{1.6}$ CFU ml^{-1} out of the initially added $10^{3.7}$ CFU ml^{-1} was obtained under competition with the native population of



Microbial Survivability, Figure 1 Seasonal changes in colony-forming marine bacteria and vibrios in Tokyo Bay. Marine bacterial populations (*closed circles*), non-sucrose-metabolized vibrios (*closed triangles*), and sucrose-metabolized vibrios (*closed squares*) were examined seasonally in 2008. Seawater samples were taken at a seawater pump in the engine room through the sea chest located at approximately 6 m under the surface of the body of water.

$10^{3.2}$ CFU ml^{-1}. On the other hand, the number of total populations was not affected, regardless of the varying numbers of *Vibrio* sp. cells in the initial samples.

McCarthy (1996) has examined the survivability of pathogenic *V. cholerae* O1 in seawater in relation to temperature and salinity. A ballast water isolate took 240 days to obtain a one-tenth reduction in the number of colony-forming cells in autoclaved seawater (32 ppt salinity) at 18 °C. While the isolate was suspended in raw seawater, the survival time was reduced to 12–27 days. These results indicate that biological competition of exotic species with the native population reduces the survivability to only a few weeks after being introduced into estuarine environments.

Summary

Nonnative and unwanted species, such as toxic dinoflagellates and pathogenic *V. cholerae*, are transported with ships' ballast water. Some of these species have become successfully established and are reproducing in the new environments. Prevention of invasive exotic marine organisms is very important for the conservation of biodiversity as well as the reduction of economic damage to fisheries and industries.

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Cross-references

Anthropogenic Impacts
Invasive Species
Phytoplankton

MICROFAUNA

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Definition

Microfauna are minute organisms typically found in high abundances in estuarine and coastal marine bottom sediments (Fenchel, 1969, 1978; Alongi, 1990; Day et al., 2012). They are not only important members of the benthos but also play a significant ecological role in pelagic environments (Fenchel, 1988, 2001). The term is largely reserved for protozoans (ciliates and foraminifera) which pass through a sieve mesh size of 100 μm (Fenchel, 1978), although other organisms are included as well (Kennish, 2001). The zooflagellates, the smallest protozoans weighing $\sim 10^{-11}$ g, are important microfauna in estuaries (Kennish, 1986).

Description

The ciliates, which range in weight from $\sim 10^{-10}$ to 10^{-6} g, are among the best studied protozoan groups (Kennish, 1986). They generally reach greatest numbers and biomasses in fine sediments and in bacterial mats of estuaries, typically attached to grain surfaces or in interstitial spaces where they often form complex communities. Many species of microfauna are known from studies of benthic ecosystems, and many others are vital to pelagic food chains and thus system function (Fenchel, 2001).

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Cross-references

Benthic Ecology
Meiofauna
Macrofauna

MICRONUTRIENTS

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Synonyms

Microminerals; Minor elements; Nutrients; Trace elements

Definition

Micronutrients are essential elements with biological functions at small concentrations (<0.01 %). They include minerals required in low quantities for the growth of organisms. In analytical chemistry, a trace element is a chemical element with a concentration of less than 100 parts per million (ppm).

Description

Importance: Minor and trace elements are required by autotrophs, although their availability in estuaries is usually not limiting to growth. In estuaries, these elements are typically available, free, or combined to meet the demand of phytoplankton. This availability can yield high levels of biomass (Day et al., 2012).

Sources: Most biologically important compounds entering an estuary are from riverine sources, in particulate or dissolved forms. When river water mixes with seawater, these compounds are retained in an estuary (Bianchi, 2007).

Species: Trace elements include iron, cobalt, chromium, copper [2], iodine, manganese, selenium, zinc, and molybdenum. Micronutrients also include vitamins, which are organic compounds that organisms require in low concentrations (Falkowski and Raven, 2007). The most commonly occurring forms of iron are the insoluble form (Fe_3^+) and the soluble form (Fe_2^+). Speciation studies have shown that dissolved copper in most natural waters is mainly organically complexed (Stumm and Morgan, 1996). By dominating copper speciation, organic complexation may control copper's bioavailability.

Demand: Microelements and trace elements are essential for autotrophic organisms; they are required as nutrients in very small concentrations. They are essential to the photosynthetic process, being key constituents of chlorophyllous pigments and enzymes. Many investigators have shown that iron (Fe) is the most important bioactive trace element. N_2-fixing cyanobacteria require larger amounts of iron than other micronutrients.

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Cross-references

Estuarine Circulation
Nutrient Limitation
Phytoplankton

MICROPHYTOBENTHOS

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Synonyms

Benthic microalgae

Definition

The microphytobenthos are microscopic autotrophs associated with benthic substrates. They are usually found as single cells but may also occur as filamentous or colonial forms. Some species are firmly attached to substrates, e.g., most monoraphic and biraphic diatoms, while others are loosely attached or not at all, or even motile, as with benthic dinoflagellates. “Epipsammic” forms are associated with sandy substrates, “epipelic” forms with muddy substrates, “epilithic” forms with rocky or other hard substrates, and “epiphytic” and “epizooic” forms with macroflora and macrofauna, respectively.

Taxonomy

The microphytobenthos are usually dominated by pennate diatoms (Bacillariophyceae, Pennales), but not exclusively so. Some centric diatoms (Centrales) are also predominantly benthic in life mode. Other autotroph taxa frequently found in microphytobenthos assemblages include dinoflagellates, cyanobacteria, chlorophytes, and euglenoids. Superior resistance of siliceous diatom frustules to damage during sampling and laboratory examination may bias taxonomic assessments of the microphytobenthos.

Ecological significance

The microphytobenthos are frequently important in estuarine ecosystems in terms of biomass, production, and trophic significance and also participate in nutrient cycling and the consolidation of sediments. Comprehensive reviews of the importance of microphytobenthos in estuarine ecosystems have been published by MacIntyre et al. (1996), Miller et al. (1996), and Underwood and Kromkamp (1999).

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Cross-references

Autotrophic
Benthic Ecology
Cyanobacteria

MINERALIZATION

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Synonyms

Conversion of organic compounds

Definition

Mineralization, as it relates to estuaries, refers to the conversion of organic compounds, and those organic compounds with N, P, S, or C, to free inorganic radicals and ions available for biotic uptake, the formation of inorganic mineral or mineral/organic inclusions and structures in plants and animals, the biomediated precipitation of minerals, the inorganic precipitation from aqueous solution of mineral substances, and the (diagenetic) inorganic transformation of one mineral to another. All of the above processes can occur in estuarine environments.

Types of mineralization

Mineralization has different meanings depending on the scientific discipline or subdiscipline where the term is being applied, and to separate them, specific adjectives or suffixes are used herein. Ecologists, biologists, and biochemists view mineralization as the conversion via biochemical breakdown, often facilitated by microbes, of organic compounds and organic compounds with molecular attachments (attached radicals, or functional groups or substituents, or ligands), such as NH_4, NO_3, SO_4, PO_4, and CO_3, to inorganic molecules and ions and the transformation of organic compounds to free C-bearing ions or C-bearing molecules. Effectively, this is the process of converting organic compounds to inorganic compounds and ions. The inorganic ions are then available for plant uptake by macrophytes in tidal and aquatic environments, by benthic diatoms and phytoplankton, or further metabolized by bacteria and fungi. This form of mineralization, while resulting in inorganic products, does not imply the development of "minerals" in the sense of geology (see Bates and Jackson, 1987). Biologists, as anatomists, view mineralization as the formation of crystalline or amorphous minerals or mineral/organic materials such as exoskeletons and endoskeletons in plants and animals, i.e., biomineralization – the formation of calcitic mollusc shells is an example. Biologists, geochemists, and geologists view mineralization as the formation of crystalline or amorphous substances as precipitates in sediment or other substrates facilitated by organisms, i.e., biomediated mineralization – pyrite framboids in anoxic sediments are an example. Geologists also view mineralization as the precipitation of minerals (inorganic elements and/or compounds) into a preexisting medium or directly onto a substrate, i.e., geological mineralization – precipitation of gypsum from water is an example. In the case of estuarine environments, geological mineralization is the formation of inorganic crystalline minerals or amorphous minerals by direct precipitation from estuarine water onto and into substrates. Geologists also view synsedimentary diagenesis or early diagenesis as a variant of mineralization, where there is chemical or crystallographic alteration of preexisting minerals into new mineral species or crystallographically altered species – the oxidation of pyrite in formerly anoxic sulfidic sediments is an example.

Conversion of organic molecules to inorganic molecules (biological mineralization)

Biological mineralization (or biotransformation) is the conversion of organic molecules to inorganic molecules and ions and can be complex and multidimensional. In fact, there are many pathways and processes subsumed under the term "mineralization" in which organic matter is mineralized by fermenting, denitrifying, sulfate-reducing, and methane-producing bacteria (Jørgensen, 1982), some under anaerobic conditions and some under aerobic conditions. One major pathway, for instance, involves the breakdown of organic molecules and their oxidation by sulfate-reducing bacteria, which utilize the sulfate exogenically in the environment as the energy source for the decomposition.

Many organic molecules may have attachments, or substituents, of NH_4, NO_3, SO_4, and PO_4, which are important for plant growth, but the N, S, and P in their organically bonded forms are not available for plant uptake. Biochemical decay, or microbial decay driven by bacteria and fungi, result in breakdown of such molecules and the production in solution of inorganic free radicals such as NH_4^+, NH_3, NO_3^-, SO_4^{2-}, PO_4^{3-} (i.e., mineralization) that are available for plant uptake. Mineralization also involves the transformation of organic compounds to free C-bearing ions or C-bearing molecules such as carbon dioxide (CO_2) and methane (CH_4). In these cases, mineralization is the transformation of organic compounds, often with attached N, P, S, and C, into plant-accessible inorganic forms. Once incorporated into plants, it is then available to be digested and incorporated into animals.

Some of the best studied mineralization pathways in the transformation of organic compounds to inorganic ions and compounds is the conversion of nitrogen, bonded as an element or part of a radicle, in proteins, amines, and nucleic acids, among others, to ammonium ions, ammonia, and nitrates (i.e., NH_4^+, NH_3, NO_3^-). The details of mineralization of organic compounds with nitrogen and the re-incorporation of nitrogen into organisms is complex and variable depending on the primary material, the environmental conditions, and the microorganisms. The processes have been assembled from studies of the nitrogen cycle (Canfield et al., 2010; Fowler et al., 2013; Galloway et al., 2013) wherein nitrogen, as atmospheric gas, is incorporated into organic compounds in plants and animals initially via nitrogen-fixing bacteria or compounded as inorganic nitrogen compounds via lightning and then cycled through various biological processes to ultimately return to the atmosphere. At intermediate stages, upon the death of a plant or animal, or when an animal expels waste, nitrogen is present in a variety of organic molecules. Bacteria, or fungi, by use of enzymes, convert the organic compound bearing the nitrogen into ammonia, becoming ammonium ions (NH_4^+) when dissolved in water. This particular process of mineralization is termed "ammonification."

The conversion of ammonia/ammonium to nitrite and nitrate is performed primarily by bacteria and other nitrifying bacteria (Kaplan, 1983; Henriksen and Kemp, 1988). In the primary stage of nitrification, the oxidation of ammonia is performed by bacteria such as *Nitrosomonas*, which converts ammonia to nitrites (NO_2^-). Other bacteria, such as *Nitrobacter*, are responsible for the oxidation of the nitrites into nitrates (NO_3^-) since it is important for the ammonia to be converted to nitrates because accumulated nitrites are toxic to plant life. Denitrification, the next step in the nitrogen cycle, is the reduction of nitrates back to nitrogen gas (N_2), which then completes the nitrogen cycle. This process is performed facultatively by bacteria such as *Pseudomonas* and *Clostridium* in anaerobic conditions, using the nitrate as an electron acceptor in the place of oxygen during respiration.

Other well-studied mineralization pathways include the conversion of organic compounds with P, such as in phytin, proteins, DNA, RNA, nucleic acids, and phospholipids, to inorganic PO_4^{3-}. The process of deriving inorganic phosphorous from phosphorous-bearing minerals (e.g., apatite) or from decaying organisms, released as PO_4^{3-} in solution, and its incorporation into organic compounds and return after death and decay of the organisms to the environment as inorganic ions or precipitated as apatite are part of the phosphorus cycle (Filippelli, 2008).

Another well-known biological form of mineralization is the oxidation of organic molecules, including those with sulfur bonded in organo-sulfur compounds (such as some proteins and amino acids, thiocarboxylic acids, thiols, thioethers), to soluble sulfate and sulfide ions (SO_4^{2-}, S^{2-}). The bacterial reduction of sulfate to sulfide in sediments, seawater, and pore water is viewed to be a key process in the oceanic sulfur cycle and is responsible for the oxidation and mineralization of the organic matter buried below the oxic and suboxic zones of sediments (Jørgensen, 1982). The process of transformation of inorganic forms of sulfur radicles and ions to organically bound sulfur, and their conversion after death and decay of the organisms to the environment as inorganic ions or even as precipitated minerals (such as sulfides), is part of the complex sulfur cycle (Canfield and Raiswell, 1999).

Formation of minerals as exoskeletons or endoskeletons (biomineralization)

Many organisms inhabiting estuaries, in environments from the riverine extreme to the marine mouth, precipitate and use inorganic minerals such as calcite, Mg calcite, aragonite, amorphous silica, magnetite, and various phosphatic minerals and compounds as their exoskeletons and as endoskeletons for protection, for strengthening, and as an aid to feeding, orientation, and navigation. Such mineralization is commonly termed by some authors as "biomineralization," i.e., the process by which living organisms produce mineral products (Simkiss and Wilbur, 1989). In the arena of biologically mediated mineral growth, and its utilization internally or externally by an organism, the distinction between "mineral" and "organic" becomes blurred. For instance, calcite precipitated from mineral-enriched waters is considered to be an inorganic mineral, but calcite is also formed by an organism as shell which is chemically, mineralogically, and crystallographically similar to the inorganic form. The problem is particularly complex where hard organic structures and skeletons formed by organisms are comprised of interlayered organic material and crystalline substances. For example, chitin, a polysaccharide, is a long-chain polymer of *N*-acetyl-D-glucosamine and, though organic in composition and origin, has a crystallographic structure (Carlstrom, 1957; Raabe et al., 2007), occurring as nanocrystallites. At the nanoscale it is a crystal and essentially an organic "mineral" (or a biomineral). Further, as an endoskeleton and exoskeleton, chitin commonly is structurally compounded/interlaced or interlayered with proteins and inorganic minerals such as calcite and phosphates. Chitin comprises the exoskeletons of arthropods such as crustaceans (e.g., crabs) and insects, the radulae of molluscs, and the beaks and internal shells of cephalopods.

Probably the best known development of minerals by aquatic organisms, including estuarine organisms, at the macroscopic scale, is the precipitation of exoskeletons of calcite and/or aragonite by molluscs, evident as the familiar shells of bivalves and gastropods (Bathurst, 1975; Rhoads and Lutz, 1980). Here, the molluscan organisms, by use of their mantle, facilitate precipitation of an exoskeleton of calcite (e.g., oysters), or of aragonite (e.g., *Paphies*, the pipi shell, or *Donax*, the bean clam), or mixed layers of calcite and aragonite. The precipitation of the carbonate minerals can be complex: it involves a sequential relationship and interactions between the animal, soft-tissue organic compounds, hard organic structures, an early amorphous stage of carbonate development, and the final stage of calcium carbonate crystal development nucleated on earlier stage crystals (Addadi et al., 2006).

Grazing molluscs, such as chitons, also can precipitate the mineral magnetite (Fe_3O_4, as a mineral hard enough to score or scratch some common rock minerals, especially calcite) in their radulae as teeth for use in scraping algae, diatoms, and biofilms off rocks. Molluscs also use apatitic calcium phosphate, or limonite, or amorphous silica in their radulae (Lowenstam and Weiner, 1989; Macey et al., 1994; Liddiard et al., 2004). More recent studies have shown magnetite, lepidocrocite, goethite, apatite, and amorphous precursor phases to be also present in some species (Saunders et al., 2009). Predatory molluscs such as Naticidae and Conidae use a chitinous radula for drilling into the carbonate exoskeletons of other invertebrate fauna (Carriker, 1961).

Plants also precipitate calcite or aragonite as crystallites in their tissue. The best known are the green algae *Acetabularia* and *Halimeda* and Charophytes (Bathurst, 1975). They precipitate these carbonate minerals as fine-grained crystals, which after death of the plant may be released into the environment as mud-sized free crystallites, contributing to the mud fraction of estuarine sediments.

At the microscopic scale, the best known development of minerals by estuarine organisms is the precipitation of crystalline calcite, Mg calcite, and aragonite by foraminifera (Bathurst, 1975) and the precipitation of amorphous silica (SiO_2) by diatoms (Kröger et al., 1999). Diatom biogenic silica is mainly amorphous, hydrated SiO_2 containing a small proportion of organic macromolecules, which are thought to control silica deposition and nanopatterning (Sumper and Kroger, 2004).

A variety of other estuarine invertebrate fauna, besides the molluscs, precipitate minerals or hard crystalline or amorphous substances, as endoskeletons or as exoskeletons, in the form of carbonate minerals (usually as $CaCO_3$), chitin, complex carbonate/phosphate minerals, or amorphous silica. These include the crustacea (chitin, calcite, and calcium phosphate; Dennell, 1960; Warner, 1977; Rhoads and Lutz, 1980; Raabe et al., 2007), other arthropods (chitin), sponges (biogenic silica), bryozoans (calcite and aragonite), and echinoderms (calcite).

Outside the realm of invertebrate fauna, vertebrate fauna in estuaries, ranging from sharks, to fish, to reptiles (e.g., crocodilians), to marine mammals, and seabirds produce endoskeletons, mineral feeding structures, and other structures of bone and teeth which, in terms of mineral content, range from calcium carbonate to calcium phosphate to magnetite (Lees, 1989; Glimcher, 2006; Pasteris et al., 2008).

Vertebrate fauna also specialize in the development of magnetite in tissue (Fe_3O_4; Kirschvink and Gould, 1981; Diebel et al., 2000) and otoliths ($CaCO_3$ in fish; Gauldie, 1993; Campana, 1999). Bone is a composite material composed of organic components (primarily collagen) and inorganic components (bio-apatite, $Ca_{10}(PO_4)_3(OH)_2$), as well as lipids and water. As a crystalline solid, bone has a special form of the mineral apatite that dominates its composite structure, being composed of extremely small-sized crystallites (Wopenka and Pasteris, 2005).

The long-term products of biomineralization of biota in estuaries is biogenic sediment through the accumulation and formation of foraminiferal sand, molluscan shell grit and gravel, diatomaceous mud, and carbonate mud.

Formation of minerals via biomediation (biomediated mineralization)

The bacterial reduction of sulfate to sulfide at the sediment bed is a key process in the coastal sulfur cycle and is responsible for the oxidation of organic matter which becomes buried below the oxic and suboxic zones of the sediment bed (Lovley and Phillips, 1986; Machel, 2001). The sulfate ion is common in seawater, sediment, or in waters rich with decaying organic material (with SO_4^{2-} deriving from organic matter). Sulfate-reducing bacteria are common in anaerobic environments wherein they utilize the sulfate ion as an electron donor and thus aid in the degradation of organic materials. Sulfate reduction is the dominant terminal step in the mineralization processes of sulfate-rich sediments where the sulfate reducers inhibit the methanogens by competing for common substrates. Sulfate reduction thus is quantitatively important in the overall oxidation of organic matter in sediments (Barton and Fauque, 2009).

Various minerals can be precipitated by biomediation, with the best known being pyrite. If Fe, Mn, Cu, and Zn are present in the environment, they can be involved in the formation of sulfides, however, since Fe is generally the most common transition metal cation in natural environments, sulfides of Fe (as pyrite) will be the most common mineral. Precipitates of pyrite, mediated by sulfate-reducing bacteria, are commonly framboidal (Sawlowlicz, 1993). Framboids (after the French word *framboise* for "raspberry") are small clusters of pyrite crystals resembling a raspberry that can be < 1 µm in size but range in size from 0.5 µm to 40 µm in diameter, with the average aggregate size about 5 µm (Wilkin et al., 1996). That FeS_2, as framboids, is associated with microbial activity has been known for some time and has been implicated in the development of sedimentary pyrite in the rock record (Schieber, 2002). While organic-matter-rich sediments inherently tend to be black or dark gray, the fine-grained precipitated iron sulfide disseminated throughout the sediment as a result of bacterial degradation of organic material typically also renders sediments to various shades of gray to black depending on the concentration of the pyrite in the sediment.

Iron sulfides as pyrite (FeS_2) and iron monosulfide (FeS) can play a central role in the sulfur and iron cycles of estuarine, coastal, and marine sediments. They may be buried in the sediment (remaining gray to dark gray) or oxidized by O_2 when brought to the near surface or surface by wave action or by bioturbation. FeS_2 and FeS may also be oxidized within the anoxic sediment in which NO_3^-, Fe (III) oxides, or MnO_2 are available as potential electron acceptors (Schippers and Jørgensen, 2002).

As a result of the shell-forming processes, organic matter may be embedded in molluscan calcitic/aragonitic shells and crustacean chitinous/carbonate exoskeletons where, if decayed by bacteria, becomes a micro-site for biomediated precipitation of fine-grained pyrite framboids. The scattered occurrence of fine-grained pyrite may impart a speckled appearance to shells and, in the extreme, may turn them gray. Also, when embedded in sulfidic sediment, shells often become blackened with pyrite through sulfide diffusion. As such, calcitic/aragonitic shells and chitinous/carbonate shells become sparsely to moderately impregnated with pyrite. Blackened shells and foraminifera have been reported by Ginsburg (1957), Pilkey et al. (1969), and Davies (1970). In each case, iron sulfide was implicated as the blackening agent.

Algae boring into shells often are micro-sites for the precipitation of aragonite, Mg calcite, or calcite in the algal thread tubules. They also are sites for pyrite precipitation. The calcitic/aragonitic shells become riddled with ultrafine carbonate-filled borings, and in the extreme, the carbonate-filled borings coalesce and form a thin carbonate envelope around the shell (Bathurst, 1975). In the case of pyrite, the fine-scale algal tubules are lined with or partially filled with pyrite, which imparts a gray external tinge to the shells.

Organisms, especially bacteria, are instrumental in facilitating the precipitation of microcrystals and nanocrystals of the calcium carbonates (calcite, Mg calcite, aragonite), dolomite (Castanier et al., 2000; Sánchez-Román et al., 2007, 2008), and phosphates (Sánchez-Román et al., 2007). Though not volumetrically important, this process results in contribution to the mineral record in estuaries.

Minerals directly crystallized from estuarine water (geological primary mineralization)

Minerals can also directly crystallize from estuarine water, either from the water column or from pore waters, if the hydrochemistry, salinity, or concentration of the open estuarine water or tidal flat pore water is appropriate. These are primary precipitates. The minerals usually involved are calcite, aragonite, siderite, gypsum, silica, iron oxide, iron sulfide, apatite, and, if the cationic chemistry and pore water chemistry are appropriate, minerals in complexes such as siderite, calcite, Fe monosulfide, and Zn sulfide/sulfate (Postma, 1982; Pirrie et al., 2000), occurring as cements, cemented layers, intra-skeletal crystal crusts, and nodules. The controls on precipitation of minerals can be complex involving salinity, the concentration equilibrium for a specific mineral (i.e., its saturation or oversaturation), temperature, confounding materials (such as clay particles), and/or presence of certain cations (e.g., Mg^{2+} inhibits precipitation of apatite), pH, Eh, and buffers (van Beusekom and de Jonge, 1997; Jordan et al., 2008).

Since estuarine waters are the mixing zone of marine water and freshwater, in many estuaries the open water salinity in specific parts of the estuary will fluctuate between freshwater and marine water (both extremes that are not conducive to the precipitation of minerals by saturation or oversaturation) or may remain in the brackish water field (also not conducive to the precipitation of minerals). The best locations for precipitation of minerals are the hypersaline estuarine high-tidal flats where evaporation and transpiration can increase the salinity of pore water to levels in excess of 60 ppt and into the precipitation fields of calcite, Mg calcite, or aragonite (> 60 ppt) as carbonate minerals or gypsum (> 120 ppt) (Logan, 1974). These minerals have been recorded in hypersaline parts of tidal flats of estuaries.

Where minerals are precipitated by pore waters, they form in the interstices of the sediment and can cement (i.e., lithify) the sediment. Surface crusts of cemented sediment, cemented layers within the sediment, or nodules (lumps) of cemented sediment can form in this manner (Pye, 1984; Semeniuk, 2010). In other situations, precipitation of minerals result in mineral crystals scattered in the sediment.

Some authors, through the use of hydrochemistry, have inferred precipitation of silica in estuaries, but evidence of the products of such direct precipitation of silica has not yet been documented.

Under acidic conditions, phosphorus may react with aluminum and iron to form minerals such as strengite and variscite (though uncommon).

While framboids of pyrite are commonly considered to be the products of bacterial activity, Wilkin and Barnes (1997a) have documented pyrite forming in the water column. In a shallow, fjord-like estuary of the Pettaquamscutt River (Rhode Island), there is a gradient with depth of oxic and anoxic and sulfidic water layers in which iron monosulfides increased with depth. Pyrite is suspended in the water column and is present only as framboids, whereas the sediments contain (dominantly) framboids as well as euhedral grains and infilled framboids of pyrite. Pyrite framboids form in the water column where there is high supersaturation with respect to pyrite and slight undersaturation with respect to the iron monosulfides, suggesting that framboid nucleation and growth occurs in the water column adjacent to, and below, the oxic-anoxic interface where supply of ferrous species, sulfide species, and suitable electron acceptors is available. Wilkins and Barnes (1997a) conclude that there is about 70 % of pyrite burial flux to the sediments via framboids forming at the oxic-anoxic interface and then settling. Infilled and overgrown framboids form diagenetically (see next section) in the sediments as a result of nucleation and continued pyrite growth on framboids that had originated above the sediment-water interface.

Sweeney and Kaplan (1973), in examining pyrite synthesis in the laboratory, found that the path of pyrite formation begins with reaction of hydrogen sulfide and dissolved iron. The first product is either amorphous iron sulfide or mackinawite with composition of FeS. In the presence of limited oxygen, this can change to hexagonal pyrrhotite, FeS. Further reaction of the above phases with elemental sulfur produces greigite, Fe_3S_4, and, finally, pyrite, FeS_2. The greigite has the morphological character of spherules. Pyrite framboids were found to form only on spherical nuclei. Thus, chemical, crystallographic, and textural results indicated the pathway of pyrite formation and compared favorably with naturally occurring pyrite in marine/coastal sediment. In these pathways, the primary product is amorphous iron sulfide or mackinawite, and the later mineralogical alteration is part of diagenesis (see next section).

Inorganic alteration of one mineral to another (geological mineralization)

While inorganic precipitation of mineral matter from aqueous environments can be viewed definitively as mineralization, geologists also view the inorganic transformation of one mineral to another by reaction, transformation, cation substitution, or crystallographic overgrowth/ modification as a variation on mineralization. In this case, the final mineral product is the outcome of chemical and/or crystallographic alteration of earlier-formed minerals. The specific process of mineral alteration in shallow water or in shallow sediment depths is one expression of a process termed diagenesis (Larsen and Chillingar, 1979). Where such alteration is occurring contemporaneously or nearly contemporaneously with sedimentation, it is termed synsedimentary diagenesis; where such alteration is occurring a short time after deposition of the sediments (e.g., 100s to 1,000s of years), it is termed early diagenesis (as compared to later diagenesis that occurs with deeper burial and groundwater alteration).

Common examples of diagenesis are conversion of calcite and aragonite (both $CaCO_3$) in Mg-rich environments to dolomite ($CaMgCO_3$), the conversion of iron sulfide in an anoxic sedimentary environment to iron oxides in an oxic environment, the conversion of iron sulfide (pyrite) to another polymorph iron sulfide (marcasite), crystallographic overgrowths of framboidal pyrite, and the phosphatization of carbonate minerals.

The alteration of carbonate minerals to dolomite has been documented by Cook (1973), von der Borch et al. (1964, 1975), and Semeniuk (2010). The phosphatization of dolomite in supratidal (hypersaline) environments has been recorded by Cook (1973). In the case of phosphate diagenesis, while phosphate may be taken up by organisms to form bio-matter and skeletons, if the phosphate anion is abundant enough, a proportion may be involved in mineral transformation in diagenesis.

The formation of phosphatic minerals (probably as carbonate fluorapatite) as a result of early diagenetic phosphatization of calcareous or clayey minerals in high-intertidal or supratidal sediments in estuarine environments by phosphate-rich pore waters where phosphate has been transported to the estuary by rivers has been recorded by Cook and Mayo (1980) in the Broad Sound estuary in Queensland.

Fine-grained pyrite embedded in the wood of mangrove stumps in anoxic coastal sediments of North Western Australia, when exposed to oxidizing conditions, is chemically altered to goethite and hematite (Semeniuk, 1980). Diagenetic marcasite overgrown on earlier pyrite has been reported from stranded organic-matter-rich Holocene estuarine mud in eastern Australia (Bush et al., 2004) – here diagenetic pyrite occurs throughout the marine to freshwater-brackish sediments, but marcasite only occurs in the upper part of the estuarine sequence (which comprises sediments accumulated under freshwater-brackish conditions), forming overgrowths on earlier-formed pyrite crystals.

Synsedimentary diagenetic crystallographic overgrowths of pyrite were recorded by Wilkin and Barnes (1997a), as mentioned earlier, in the estuary of the Pettaquamscutt River. Framboids of pyrite that have settled on the sediment surface and have been shallowly buried are infilled and crystallographically overgrown, locally to have euhedral overgrowths as a result of nucleation and continued pyrite growth on framboids originally formed above the sediment-water interface.

As an example of synsedimentary diagenesis, Clark and Lutz (1980) reported that living molluscs in a tidal marsh are being pyritized (not by framboids but by euhedral pyrite crystals), indicating that pyritization can commence in a shell even before the death of the organism. In one shell Clark and Lutz (1980) found pyrite replacing aragonite in parts of its outer shell layer, especially around fractures, borings, and concentrations of organic matrix.

Summary

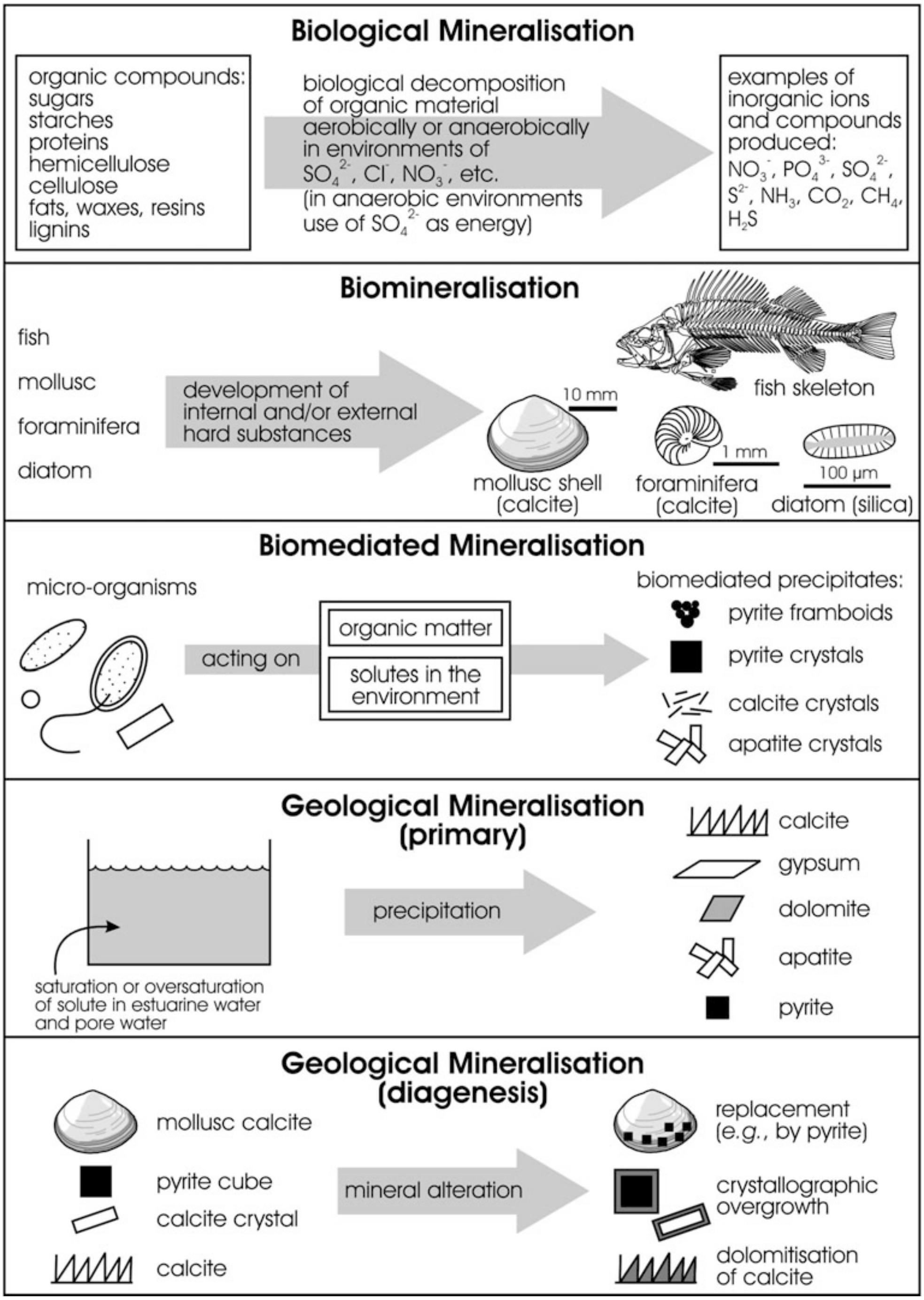
In estuarine environments there are five types of mineralization (Figure 1). The most prevalent is the transformation by biochemical breakdown, including by microorganisms, of organic molecules of plants and animals to inorganic molecules such as PO_4^{3-}, SO_4^{2-}, S^{2-}, NH_4^+, NH_3, NO_3^-, and NO_2^- and the oxidation of organic matter. With these processes, free molecules and ions are available for plant uptake or for further chemical processing. Mineralization specifically by sulfate reduction, resulting in the widespread oxidation of organic matter in estuarine sediments and in precipitation of fine-grained pyrite as a mineral product, quantitatively is also an important process. Estuarine environments accumulate organic matter in their sediments, and sulfate-reducing bacteria in anaerobic environments utilizing sulfate ions to aid in the oxidation and degradation of this organic material produce sulfide which, in combining with any available Fe, produces iron sulfides, including FeS_2. As such, pyrite as fine-grained framboids, formed through biomediation of sulfate-reducing bacteria, is a common and widespread product in estuarine sedimentary deposits.

There is a range of other minerals that form in estuarine environments. Many are biominerals formed by organisms as skeletal structures for support, defense, tissue strengthening, feeding apparatus, or navigation; these include calcite, aragonite, bone, and magnetite. Upon death of the organism, these biominerals can be incorporated into the sediments of estuaries. In this regard, this aspect of biomineralization contributes to the sediment accumulation in estuaries, from gravel-sized particles to mud-sized particles.

Some of the minerals in the estuarine environment are direct inorganic precipitates from estuarine waters and accumulate on the sediment surface or within sediment interstices, the latter potentially cementing estuarine sediments. Some of the more common minerals include calcite, aragonite, gypsum, apatite, and pyrite. A range of other minerals are diagenetic, formed as alteration products of earlier-formed minerals; these include goethite and hematite as products of the oxidation of pyrite, crystallographic overgrowths of framboidal pyrite, marcasite formed by alteration of pyrite, dolomite formed by alteration of calcite/aragonite, phosphatization of carbonate minerals or clay minerals, and the pyritization of molluscan carbonate.

The various minerals, their diagenetic alterations, and biominerals often are specific to sub-environments in the estuary. For instance, mineral precipitates can be environment specific in their occurrence: freshwater seepage is the specialized location for shoreline primary carbonate precipitation and for diagenetic formation of dolomite; hypersaline tidal flats and tidal zones are the specialized location for carbonate precipitation as calcite, aragonite, and gypsum; dark gray sandy and muddy sediments, abundant with fine-grained pyrite framboids, are ubiquitous in estuaries wherever there is accumulating organic matter and anaerobic conditions; and as such, this form of pyrite mineralization is widespread in estuaries. In terms of biominerals, leaving aside the factor that biogenic skeletal materials can be fragmented and transported, often environment-specific biotic assemblages result in skeletal suites with either diagnostic shell types and their fragments (Semeniuk et al., 2000) and possibly mineralogy, or a specific proportion of foraminiferal tests and diatom frustules (Revets, 2000), or vertebrate skeletal parts, all with their geochemical/mineralogical signatures related to sub-environments in the estuary.

Interesting minerals that form in anoxic estuarine environments are pyrite framboids and, while they have been much studied, the details of their formation are still in debate (Wilkin and Barnes, 1997b). At one extreme they are viewed to be wholly biogenic, i.e., pyritized bacteria (Schneiderhohn, 1923; Love, 1957), or pyritized organic spherical globules or gaseous vacuoles



Mineralization, Figure 1 Idealized illustration showing the five types of mineralization in estuarine environments, ranging from the biological production of inorganic compounds and ions from organic substances facilitated by microbial decay and transformations (biological mineralization), the biological production of mineral substances within and external to living organisms (biomineralization), the biologically mediated production of mineral substances (biomediated mineralization), and the inorganic precipitation of minerals from water (geological primary mineralization) to the alteration of existing minerals (geological mineralization as diagenesis). Only selected examples of each of the various mineralization types and their products are shown.

(Kalliokoski and Cathles, 1969; Rickard, 1970), and at the other extreme, totally inorganic, based on their synthesis in the laboratory and occurrence in igneous rocks and ore deposits (Berner, 1969; Love and Amstutz, 1969; Farrand, 1970; Sunagawa et al., 1971; Sweeney and Kaplan, 1973; Ostwald and England, 1979; Graham and Ohmoto, 1994). Reviews of the different genesis of framboids are presented by Love and Amstutz (1966), Rickard (1970), Kalliokoski (1974), and Wilkin and Barnes (1997b), and a recent review and description of crystallography, structure, and origin is provided by Ohfuji et al. (2005).

Classifying precipitates as either biomediated mineralization or geological mineralization can be difficult. Often it is not clear to which category the mineralization belongs, as is the case for the precipitation of carbonates and sulfides. The complexities of sulfide precipitations are highlighted by Machel (2001). In a review of sulfate reduction mechanisms (which would result in the formation of pyrite framboids), Machel (2001) concluded that the association of dissolved sulfate and hydrocarbons is thermodynamically unstable in nearly all low-temperature environments and that redox reactions would occur whereby sulfate is reduced by hydrocarbons either bacterially (bacterial sulfate reduction) *or* inorganically (thermochemical sulfate reduction). As such, their geological products are similar. Machel (2001) also concluded that in addition to iron sulfides, both galena (PbS), and sphalerite [(Zn,Fe)S] can form as by-products of hydrogen sulfide generation, if the appropriate transition or base metals are present or transported to a bacterial sulfate reduction site or a thermochemical sulfate reduction site. In this context, generation of metallic sulfides can be part of the bacterial sulfate reduction process (and be classified as biomediated mineralization) or produced inorganically in the appropriate estuarine environment (and be classified as geological mineralization). In addition, once they are precipitated, the sulfide crystals can be subject to secondary alteration crystallographically and mineralogically (i.e., diagenesis). The description by Wilkin and Barnes (1997a) of the precipitation of iron monosulfides, and various other mineral phases of iron sulfide and their transformations, for instance, indicates that the distinction between geological primary mineralization and secondary alteration (diagenesis) of the primary precipitates can be complicated.

Mineralization in estuaries is manifest at several scales. At the largest scale, the scale > 1 cm, it is expressed as biomineralization within the shelly invertebrates and vertebrate fauna and as geological mineralization and diagenesis in the larger crystals, crusts, and nodules that may be precipitated on tidal flats. At the next scale, the scale ~ 1 mm and less, it is expressed as interstitial precipitates in geological mineralization and diagenesis and as biomineralization in the exoskeletons, tests, and frustules of meiofauna, foraminifera, and diatoms. At the finest scale, the scale of 1–100 μm (or less), it is expressed as pyritic framboids and as carbonate crystals derived from disintegrated calcareous algae or from bacterially mediated mineral precipitates. In this range, though representing the finest scale of mineralization (in the sense of precipitation of mineral matter), it is the pyritic framboids that are the most widespread mineralization phenomenon in estuaries.

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Cross-references

Biogenous Sediment
Diagenesis
Microbial Degradation
Nitrification
Nutrient Dynamics
Shell Beds

MIXOTROPHIC PLANKTON

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Definition

Mixotrophic plankton are those that use a combination of phototrophic and heterotrophic nutrition.

Mixotrophic plankton and our evolving view of pelagic foodwebs

Traditionally, oceanographers and limnologists categorize plankton as "phytoplankton" and "zooplankton," implying functional roles analogous to plants and animals, respectively, in terrestrial systems. However, particularly among the protistan plankton of the lower trophic levels, the distinction between "phytoplankton" and "zooplankton" is not clear-cut. Mixotrophic nutrition among plankton has been conceptualized as a range of alternative nutritional strategies that fall somewhere between pure photoautotrophy and heterotrophy (Jones, 1994), including the auxotrophic requirement of most phototrophs for organic cofactors (vitamins), the saprophytic uptake of organic molecules by phototrophs, phagotrophic ingestion of prey by phototrophic species, and the stealing of chloroplasts from prey species that is referred to as kleptoplastidy. It has become clear that such a classification would include nearly all protistan plankton, so recently the term "mixotrophic plankton" has been reserved for those species using a combination of photosynthesis and phagotrophy (Flynn et al., 2012). Stoecker (1998) delineated three functional types (model I, II, and III) of phagotrophic mixotrophs based on functional responses to prey and nutrients and on the relative roles/predominance of heterotrophy and autotrophy, functional responses of grazing to light and nutrients, and the possible benefits of mixotrophic nutrition to the organisms. An "ideal" mixotroph (model I) is equally adept as either an autotroph or a heterotroph. Phagotrophic "algae" (model II) are primarily photosynthetic, but ingest prey under certain conditions. Photosynthetic "protozoa" (model III) are primarily phagotrophic, but can photosynthesize under certain conditions using either their own plastids or those they sequester as "kleptochloroplasts" (Stoecker, 1998). Mixotrophic plankton are widespread in estuarine ecosystems, but because of the inherent metabolic and functional diversity within this group (both within and between species), their impact on pelagic food webs is challenging to estimate (Stickney et al., 2000; Flynn et al., 2012), as are the potential effects of climate change on the role of mixotrophic plankton in pelagic nutrient and energy cycling (Wilkens et al., 2012).

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Cross-references

Phytoplankton
Phytoplankton Blooms
Zooplankton

MOSQUITO DITCHING

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Definition

Mosquito ditching is a method of controlling mosquito breeding in wetlands by draining them.

Description

Mosquito control by ditching drains standing water off of coastal wetlands to reduce the natal habitat for mosquito larvae while simultaneously allowing aquatic predators of mosquito larvae to access their prey (Smith, 1904). Mosquito ditches were widely installed in tidal wetlands in the eastern United States in the 1930s, when Depression-era work relief programs were assigned the task. In New England, over 90 % of salt marshes have mosquito ditches (Crain et al., 2009) (Figure 1).

Mosquito Ditching, Figure 1 Mosquito ditches in Rumstick Marsh, Barrington, RI. (1998 orthophotograph from RIGIS, http://www.edc.uri.edu/rigis.).

Drainage from mosquito ditches has altered marsh hydrology and community structure. Ditched marshes have fewer and smaller ponds than unditched marshes (Adamowicz and Roman, 2005), fewer waterlogged forb pannes (Ewanchuk and Bertness, 2004), and more high marsh shrubs such as *Iva frutescens* (Bourn and Cottam, 1950). Wading birds prefer non-ditched marshes, where there are more ponds (Clarke et al., 1984).

Mosquito ditches can have impacts decades after their installation as a result of interactions with other stressors. In Cape Cod, mosquito ditch banks provide pathways of *Spartina alterniflora* that are exploited by *Spartina* specialist crab *Sesarma reticulatum*. In abundance, *S. reticulatum* herbivory causes marsh die-off. Where ditches occur in combination with heavy recreational fishing that reduces the abundance of *S. reticulatum*'s predators, larger die-offs occur (Coverdale et al., 2013).

In recent years, mosquito ditching has been supplanted by a more integrated approach to mosquito management termed Open Marsh Water Management (OMWM), which attempts to balance ecological concerns with pest control and includes restoration of mosquito ditches. OMWM has mixed and unexpected effects on invertebrate and fish populations (James-Pirri et al., 2012).

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Cross-references

Coastal Wetlands
Estuarine Deltaic Wetlands
Saltmarshes

MUTUALISM

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Definition

Mutualism is defined as reciprocally beneficial interaction between species (Hay et al. 2004).

Mutualistic interactions include the exchange of resources and services, such as metabolites, assistance in inhibiting competitors or predators, and reducing susceptibility to physical stress from wind, water flow or light (Stachowicz, 2001). For each partner, the investment in such interactions is outweighed by the benefits gained. Direct and indirect mutualisms are widespread in aquatic environments and conditional rather than obligate (Hay et al., 2004). Mutualisms in which the participating organisms live in close physical association to each other are called symbioses.

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Cross-references

Symbiosis

N

NEKTON BEHAVIORAL ECOLOGY

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Definition

Nekton refers to mobile organisms which are able to swim or otherwise move independently of all but the strongest water currents; the term usually refers to fishes, shrimps, crabs, and squids, regardless of whether they are slow-moving epibenthic or faster swimming pelagic organisms. Marine mammals and turtles are sometimes considered nekton.

Behavioral ecology is the study of the behavior that underlies the organization, distribution, and ecology of populations and communities of fishes and motile macroinvertebrates in estuaries. This deviates somewhat from definitions that focus on the evolutionary basis for behavior and the consequences of behavior for the fitness of organisms.

Introduction

Estuarine nekton assemblages are both spatially and temporally diverse, largely because of the variation and the variability in geomorphological, hydrological, and environmental characteristics of coastal systems. Central to the well-being of estuarine nekton is their ability to make physiological adjustments and to move toward more suitable environmental conditions. Both responses are usually involved in the selection and occupation of habitat for feeding, reproduction, and refuge from predators.

Senses underlying behavioral responses

Sensory modalities such as taste, smell, sight, hearing, and touch determine the activities of nekton. The importance of the various senses differs among species and often among life stages of the same species (Kotrschal, 1999). Fishes with limited capacity to move far or fast tend to be thigmotactic and cryptic and have high fidelity and a relatively small home range. Blennies, gobies, and other demersal fishes can learn the topography of an area and even return to it if displaced (Gibson, 1999). Other fishes have well-developed senses of chemical and sound detection which enable them to locate food, mates, and natal spawning areas over long distances. Insights into the reproductive behavior of fishes have come from the study of sound production and the use of passive acoustic technology. Field studies have revealed locations, timing, and factors influencing the aggregation of estuarine fishes (especially members of the drum family) for spawning, and this information has been useful to fisheries management. Vision is particularly important to predacious fishes. Differences in food preferences, foraging time, and capture behavior enable many species to co-occur within habitats. Differences in physiological tolerances, sensitivity, and mobility enable species to exploit the full range of estuarine habitats and environmental conditions.

Movement

Movement is a primary manifestation of nekton behavior. It is necessary for procuring food, growth, survival, and completing life cycles. Understanding the behavior of nekton is essential to understanding the ecology of individuals, populations, and communities as well as the roles nekton play in ecosystem function. For example, life history strategies vary widely for estuarine nekton with many species undergoing major ontogenetic shifts in motility,

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feeding, habitat requirements, and social behavior. Between larval and adult stages, some species move large distances between the ocean and rivers. In some cases, these movements are associated with spawning, but long-distance migrations of subadult year classes are also widely recognized. For other species, individuals complete life cycles within tens of meters of where they originated. Nevertheless, these individuals must also move to feed, find mates, seek shelter, and avoid adverse environmental conditions. In estuaries, nekton assemblages include species which complete their life cycles within the system and many other species which move between the rivers, estuary, and ocean at various stages of their development. Within tidal systems, most nekton regularly move between flooded intertidal and adjacent subtidal areas and along salinity, thermal, and other environmental gradients (Kneib, 1997).

Quantifying behavior and understanding ecology

Some aspects of nekton behavior including the determination of swimming speeds, feeding, spawning, and responses to environmental gradients have been revealed in studies of captive animals in tanks. In some cases, artifacts caused by captivity, containers, and a lack of a full spectrum of environmental factors are largely overcome, and observed behavior likely mimics that in nature. Direct observations of behavior in natural waterways are preferred, but field-based research on the movement of nekton in estuaries is hampered by typically turbid conditions which limit direct observations and the use of video cameras. Most of what we know about the behavior of nekton in estuaries is inferred from the analysis of collections made with nets and other devices which only capture snapshots of some of what is present at a particular place and time. Quantifying nekton with nets and traps is a challenge. Differences in swimming capacities among fishes render no single collection gear totally effective for all taxa and life stages that may be present (Rozas and Minello, 1997). Avoidance, escape, and retention are uneven and difficult to measure in towed as well as passive collection gear. Understanding the behavior of nekton taxa which have an affinity for structure, an ability to bury themselves, or a patchy distribution due to schooling is necessary for interpreting the results of studies with towed and passive collection gear.

Mark-recapture studies using traditional tags have been useful in determining the net movement of individuals between two points, and time at large can provide information on age and growth. Movements can be tracked using networks of detectors for passive integrated transponder (RFID PIT) tags. Battery-powered acoustic (radio) and satellite-linked tags allow tracking of movements of large individuals, often over large distances (McKenzie et al., 2012). Sonar has been used for decades to mark the locations and estimate abundance, biomass, and relative sizes of fishes in the water column. More recently, dual-frequency sonar instruments that can generate almost video-quality image streams of both fishes and macroinvertebrates in situ have provided the best records of nekton movements and other behaviors (Rakowitz et al., 2012). The instruments can operate in the most turbid conditions during day and night, and adjustments to the instruments can be made based on real-time viewing of the output on site. Simple viewing of the recordings provides information comparable to direct observations (e.g., by a diver if the water is clear) on distributions and interactions of animals within the viewing field, but post-collection analysis with advanced software can provide much more information on length frequency distributions, biomass, school shape, swimming speeds, turning frequency, and other metrics that cannot be quantified by other means. This technology continues to advance and provide insights into the behavior of nekton. Multibeam sonar has already revealed actions and patterns that dismiss some assumptions about the activities of estuarine nekton based on decades of data from net collections.

Summary

Observing, quantifying, and interpreting nekton behavior remain difficult but are important for informing policy for the protection and management of nekton and their habitats. Knowledge about the responses of fishes and motile macroinvertebrates to environmental cues is fundamental to understanding distributions, migrations, trophic structure, reproduction, larval transport, and strategies for habitat restoration, conservation, and fishery harvests. Advancing technology in the areas of imaging and acoustics will likely advance the quantity and quality of information on the behavioral ecology of nekton.

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Cross-references

Ichthyofauna
Lobster Migration

NEURAL NETWORKS

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Synonyms

Artificial intelligence; Artificial neural networks; Computational intelligence; Evolutionary algorithms; Fuzzy logic; Machine learning; Multilayer perceptron

Definition

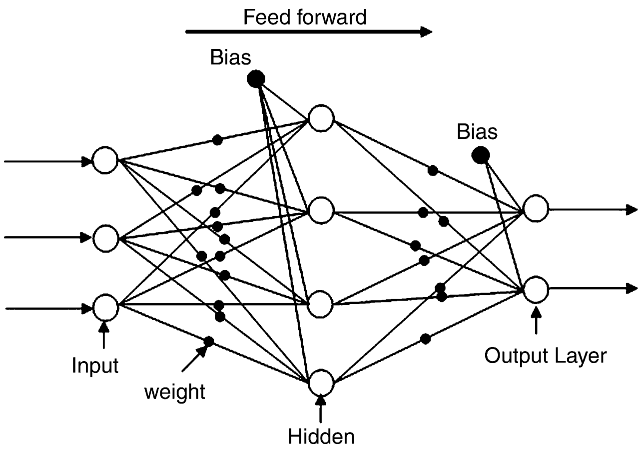
An artificial neural network, alternatively called neural network for simplicity, is an interconnected structure of computational elements called neurons or nodes.

A neural network is designed to mimic the cognition process followed by biological neurons in human brains. It is a data-driven technique and can be classified variedly under categories of soft computing, machine learning, and artificial intelligence techniques. In neural networks the intelligence results from communication between different computational elements called neurons.

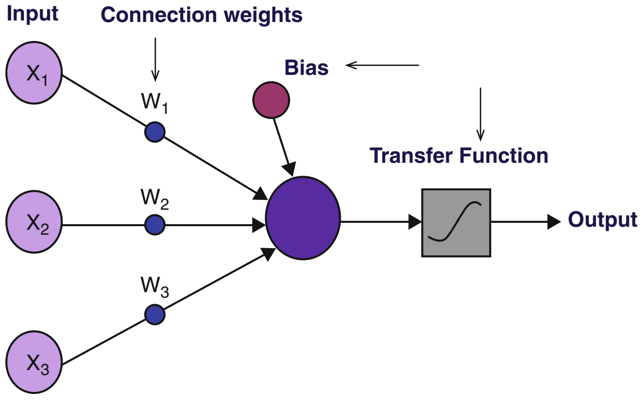
Use: A neural network is basically suited to map any random input vector with corresponding output vector, and its engineering applications include function approximation, optimization, system modeling, and pattern recognition. There are a number of advantages that the networks carry over the traditional empirical, analytical, numerical, or statistical approaches. Important among them are as follows: (1) physics or mechanics of the underlying process need not be known beforehand, (2) no a priori assumptions on data behavior are required, and (3) no exogenous input other than input-output patterns is called for to calibrate the network (Kosko, 1992).

Functioning: A typical neural network (exemplified in Figure 1) represents interconnection of computational elements called neurons or nodes, each of which (Figure 2) basically carries out the task of combining the input, determining its strength by comparing the combination with a threshold (or alternatively passing it through a nonlinear transfer function), and firing out the result in proportion to such a strength (Wasserman, 1993; Wu, 1994). Common network architecture is of feed forward type that consists of an input layer, one or more hidden layers, and an output layer of neurons.

Network Training: Before its actual application, the network is calibrated or trained. This is aimed at determining the connection weights, bias, or similar internal network parameters. In a supervised type of training, examples are used to train a network using some mathematical algorithm and until generalization results and the strengths of interconnections (or weights) are accordingly fixed, while in an unsupervised training, an input can be classified into clusters by some rule and such clusters are further used to obtain the desired outcome.



Neural Networks, Figure 1 Typical feed forward network.



Neural Networks, Figure 2 Working of a neuron.

Applications: Neural networks are used as tools to carry out a variety of tasks such as regression, optimization, function approximation, system identification, and pattern matching. Significant applications in estuarine science and engineering started in the early 1990s. A large number of past applications of neural networks in estuary-, coast-, and ocean-related studies involved estimation or forecasting of parameters characterizing environmental processes or derivation of information required for engineering activities. The environmental parameters included wave height, period, directional characteristics, spectral shapes, wave propagation, transmission, run-up, swell heights, tidal levels and time of high and low waters, sea levels and their variations, wind, currents, sea surface temperature and other met-ocean parameters, estuarine characteristics, structural damage indicators, ship design parameters, barge motions, scour depths, and soil liquefaction-related knowledge. A review of such applications can be found in Jain and Deo (2006) and Deo (2010).

Summary

Neural networks represent a computational method inspired by biological neurons of a human brain.

It consists of a set of artificial neurons each carrying out the same basic computation and linked with other neurons through certain connection paths. Before its application, the network parameters are determined through a mathematical training process for a given set of input and/or output data. Estuarine applications include estimation or prediction of parameters describing processes of tides, currents, waves, wind, sediment transport, and their interactions.

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Cross-references

Climate Change
Shoreline Changes
Tidal and Nontidal Oscillations
Tidal Hydrodynamics
Water Quality

NEUSTON

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Definition

Organisms living at the air/water interface of freshwater, estuarine, and marine habitats or referring to the biota on or directly below the water's surface layer.

Description

The neuston (Greek: *neustos* – swimming) populations of the surface layer are divided into two groups. The epineuston are organisms living on the water's surface in contrast to the hyponeuston which are organisms within a region of specified depth directly below the surface layer. The term neuston was first used by Naumann (1917) to describe species associated with the surface layer of freshwater habitats, with Zaitsev (1971) later identifying neuston composition in marine waters. These populations would include microscopic species, plus various plant and animal taxa (e.g., phyto- and zooplankton) living in this region. Gladyshev (2002) has further characterized the major physical and chemical dynamics of the surface layer influencing the composition and relationships with various neustonic populations. Investigations of these surface populations from the often more placid freshwater lake and pond habitats to the turbulence common to surface waters of estuaries and oceans have led to different interpretations regarding what populations should be included in this category and how they may be collected. Historically different methods have been established for sampling organisms occupying either the epineuston or hyponeuston regions. For descriptions and comparisons of the various collection devices, see Gladyshev (2002) and Marshall and Burchardt (2005).

Species of the estuarine hyponeuston typically come from the existing plankton populations that are in the water column and also include migratory taxa that may enter this region. The representation of these populations will also be subject to seasonal and diurnal changes in composition and abundance and will be subject to the influence of existing environmental conditions. A major sampling difference involves the hyponeuston and the depth the water sample is taken. The marine and estuarine sampling gear for the hyponeuston will typically collect a sample at a deeper depth (e.g., to 5 cm by Zaitsev, 1971, within 1 m by Hardy, 1997) than those taken in freshwater studies which generally involve a more shallow range of several millimeters (Marshall and Gladyshev, 2009). Thus, various studies of the neuston may involve different methodologies and instrumentation in sampling both the epineuston and hyponeuston biota.

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Cross-references

Phytoplankton
Zooplankton

NITRATE REDUCTION

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Synonyms

Denitrification

Definition

Nitrate reduction is the transformation of nitrate to ammonia through bacterial respiration under anaerobic conditions.

Description

Nitrate (NO_3^-) is transformed by bacteria in estuaries into reduced compounds (Day et al., 2012). Bacterial respiration is an oxidation-reduction reaction or redox reaction. Nitrate reduction takes place in terrestrial, estuarine, and marine environments and can be limited by nitrate availability. Much nitrate reduction occurs in estuarine bottom sediments (Kennish, 1997).

Nitrate-reducing bacteria use nitrate as an electron acceptor to anaerobically oxidize organic matter, which releases ammonia. This process begins with oxygen depletion (<0.5 mg/L), and nitrate becomes the primary oxygen source for the bacteria. Nitrate acts as a substitute source in the water column and anoxic sediments when oxygen consumption exceeds production or transportation, and sufficient levels of nitrate are available. Nitrate-reducing bacteria are facultative anaerobes able to exist with or without the availability of dissolved oxygen (Bianchi, 2007).

Large fluxes of (NO_3^-) from the water column to bottom sediments may be evident in areas where extensive agricultural fertilization in watersheds has led to increased rates of nitrate runoff and accumulation in estuarine water bodies (Kennish, 1997).

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Cross-references

Anaerobic Environments
Anoxia, Hypoxia, and Dead Zones
Anthropogenic Impacts
Nitrification
Nitrogen

NITRIFICATION

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Synonyms

Nitrogen cycle

Definition

Nitrification is a step in the nitrogen (N) cycle in which the most reduced forms of nitrogen are transformed by bacteria into the most oxidized nitrogen compounds. This is shown by the aerobic process where NH_3 or NH_4^+ is oxidized to NO_2^- or NO_3^- through two energy-producing reactions (Bianchi, 2007).

Description

Nitrification is a microbial process by which reduced nitrogen compounds (primarily ammonia) are sequentially oxidized to nitrite and nitrate. It is a critical step in the nitrogen biogeochemical cycle and can be modified by human activities (e.g., ammonia is available from nitrogenous organic matter degradation or input from agricultural activities; nitrate is widely used in fertilizers). Nitrification is a two-step process. Transformation of ammonia and ammonium into nitrite by bacteria of the genus *Nitrosomonas* is the first stage. Next, bacteria of the genus *Nitrobacter* complete the conversion of nitrite to nitrate (Day et al., 2012). Nitrate is the end product of organic N oxidation and is the most stable N compound in aquatic ecosystems. This process begins when the amino group ($-NH_2$) is transformed into ammonia (NH_3) – ammonification (Bianchi, 2007). The reactions are generally coupled and proceed rapidly to the nitrate form. These reactions are rapidly completed, which explains why nitrite concentrations are typically low, except in polluted areas.

Nitrogen is considered the limiting nutrient that controls primary production in most estuarine systems. Nitrogen limitation leads to a reduction in growth and photosynthetic rates (Falkowski and Raven, 2007). The bacteria responsible for aforementioned transformation are nitrifying and strictly aerobic, which generates a high demand for dissolved oxygen to complete the process. Thus, this process only occurs under aerobic conditions. It is a very important process for the balance of the trophic web; therefore, different methods have been used to assess the intensity of nitrification and abundance of nitrifying bacteria in estuaries.

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Cross-references

Aerobic Environments
Microbial Degradation
Nitrogen
Water Quality

NITROGEN

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Synonyms

Azote; Limiting nutrient

Definition

Nitrogen is the seventh element in the Periodic Table of the Elements. It is an important nutrient for estuarine and marine life and is essential for the growth of aquatic plants.

Symbol: N, Atomic number: 7, Atomic weight: 14.0067.

Description

Nitrogen is an essential nutrient for synthesizing amino acids and, consequently, proteins, enzymes, and cell wall structural components.

Forms: Nitrogen (N) is found as a dissolved salt or in gaseous form. The elemental form (N_2) is the dominant form present in the atmosphere, comprising approximately 80 % of the total atmospheric gases by volume. In its free form, N is the primary component of air. Ammonia (NH_3), ammonium (NH_4^+), nitrite (NO_2^-), and nitrate (NO_3^-) are the principal dissolved inorganic nitrogen (DIN) forms in coastal systems (Kennish, 1997; Bianchi, 2007).

Sources: Nitrogen enters estuaries through river flow, atmospheric deposition, and biological fixation.

Demand: The inorganic N forms preferred by aquatic plants are ammonia and nitrate, and few species (cyanobacteria and other bacteria) can take advantage of N_2 or organic N forms. Nitrogen is the primary limiting factor for estuarine primary production (Day et al., 2012). Nitrogen limitation leads to a reduction in plant growth and photosynthetic rates (Falkowski and Raven, 2007).

Isotopes: Only two nitrogen isotopes are stable: ^{14}N and ^{15}N. The most abundant is ^{14}N (99.634 %). An increase in ^{15}N isotope has been observed with increasing trophic level (Bianchi, 2007).

N Cycle: The biogeochemical cycle is complex and has two stages. One is external, wherein nitrification, denitrification, and ammonification processes are performed through bacterial activities. Nitrification is an aerobic process and produces the most oxidized nitrogen compound, nitrate, whereas ammonification transforms amine radicals into ammonia and free or molecular nitrogen. The other stage is internal, wherein ammonium is transformed into amine radicals and amino acids during photosynthesis (Day et al., 2012).

N_2 Fixation: This is the main source of inorganic nitrogen in oligotrophic waters. It is catalyzed by the enzyme nitrogenase, which is found only in some species of bacteria and photoautotrophic cyanobacteria (Bianchi, 2007). Only a few phytoplankton carry out N_2 fixation.

Human Impacts: The nitrogen cycle is particularly affected by human activities in estuaries. Increased anthropogenic N input to rivers and other influent systems from agricultural, industrial, and urban activities intensifies the estuarine eutrophication process, stimulating the growth of unwanted aquatic plant species. Nitrate excess from agricultural activities is transported by rivers, causing the eutrophication of many estuaries worldwide (Kennish, 1997). This process can change the trophic web and can reduce estuarine biodiversity. Environmental monitoring indicates that the global nitrogen load to the environment has doubled since the beginning of the preindustrial era. Nitrogen pollution in aquatic ecosystems can lead to the development of dead aquatic zones, such as in the Gulf of Mexico (USA). There is a consensus that nitrogen pollution of aquatic systems has reached a global impact.

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Cross-references

Cyanobacteria
Ecological Stoichiometry
Eutrophication
Macronutrients
Nitrogen

NONPOINT SOURCE POLLUTION

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Synonyms

Diffuse pollution

Definition

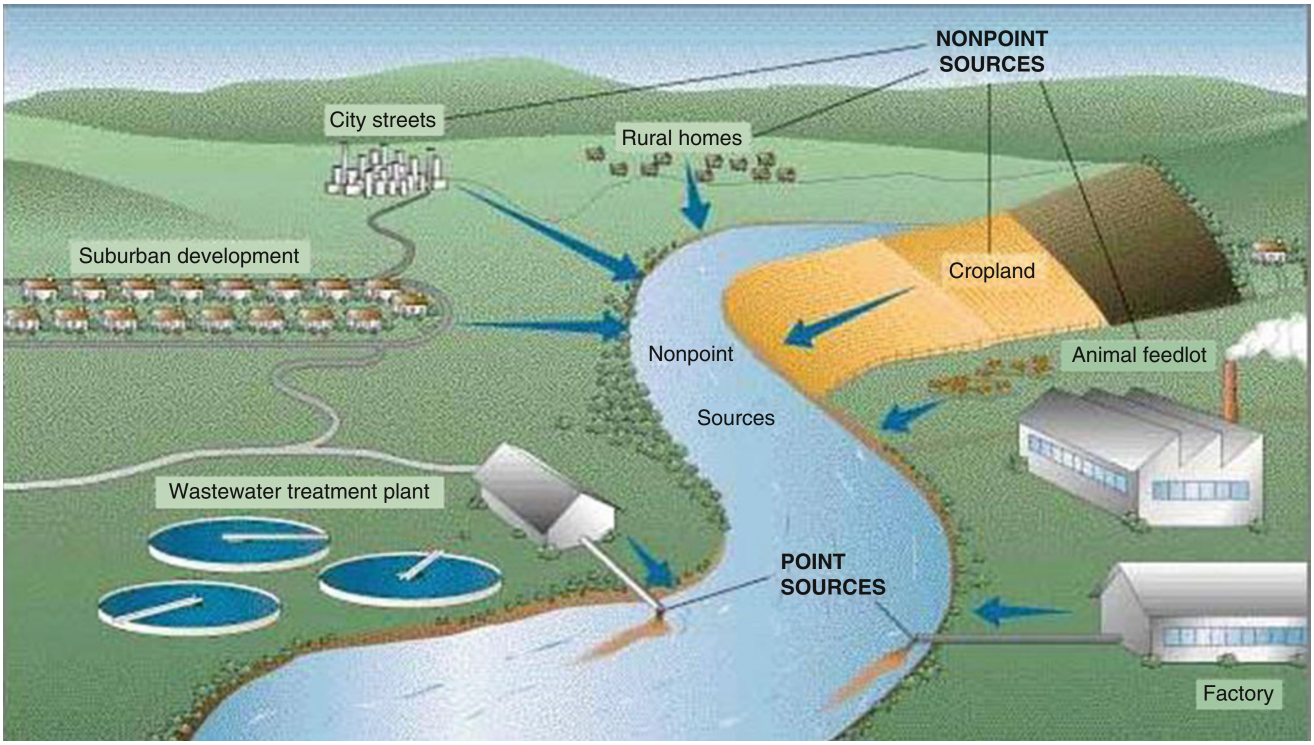
Diffuse source of pollution to receiving waterbodies, most often in the form of runoff from adjacent land or atmospheric deposition.

Introduction

Industrial discharge of chemicals, oil spills, raw sewage, and illegal dumping are the images that come to mind when most people think of the causes of water pollution. However, the Clean Water Act and related implementation of state and local regulations have been largely successful in eliminating or minimizing this type of discharge to waterbodies, and water quality improvements have been significant. Yet, water quality standards are still not being met in many waterbodies across the country. Today's largest threat to water quality in the USA comes from sources that are much more diffuse and harder to define or control. Pollution reaches waterbodies from a large variety of sources, but they can generally be categorized into two groups: point and nonpoint sources (Figure 1). Point sources are made up of discrete sources of water such as industrial outfalls or wastewater treatment plants. The specific definition from the Clean Water Act includes "any discernible, confined and discrete conveyance, including but not limited to any pipe, ditch, channel, tunnel, conduit, well, discrete fissure, container, rolling stock, concentrated animal feeding operation, or vessel or other floating craft, from which pollutants are or may be discharged. This term does not include agricultural stormwater discharges and return flows from irrigated agriculture" (Clean Water Act Sec. 502(14)). Since 1990, point sources have also been defined in a regulatory context to include municipal separate stormwater systems (MS4s) that collect stormwater runoff from urban areas. These sources are all regulated through National Pollution Discharge Elimination System (NPDES) permits issued by state governments or the USEPA. The second major category of water pollution sources is nonpoint source pollution and is composed of all other sources. Pollutants from nonpoint sources originate on surfaces like agricultural fields, parking lots, roofs, smokestacks, streets, and lawns and make their way into waterbodies through atmospheric deposition, rain, snow, or wind. Due to the diffuse nature of these sources and pathways, all types of pollutants, including sediment, oil and grease, nutrients, trash, and toxic pollutants like mercury and pesticides, make up nonpoint source pollution. The diffuse nature of nonpoint source pollution also makes it incredibly challenging to control and is part of why at least 50 % of the nation's water quality problems stem from nonpoint sources (Oberrecht, 2002).

Impacts on estuaries

Estuaries are particularly vulnerable to nonpoint source pollution for a number of reasons. Due to their location



Nonpoint Source Pollution, Figure 1 Examples of point and nonpoint source of water pollution (Source: Nebel and Wright, 1999).

and morphology, estuaries act as natural sinks for the watershed that they drain. Their location on the coast at the intersection of fresh- and saltwater provides a low-gradient environment with slow-moving water that gradually gathers runoff from the entire watershed. The low water velocity in estuaries enables the heavier sediments, and the contaminants bound to them, to settle. Some sediment is washed out to sea by tidal and current forces, only to get forced back upstream with the change in tides and winds. The wide, relatively flat morphology of estuaries encourages vegetation growth such as algal mats and eel grass beds that provide additional places for contaminated sediments to deposit. Lastly, salty water neutralizes the charge of particles suspended in the estuarine water column, and they are better able to clump together and settle to the bottom (Oberrecht, 2002). In addition to natural reasons of vulnerability, the coastal areas of the USA that are home to estuaries are also the most densely populated. In 2011, the population density of US coastal counties was over four times that of the average nationwide density (U.S. Census, 2011). Increased population places particular pressure on estuarine water quality since population growth is linked to increased human activities that create nonpoint source pollution (Bricker et al., 1999).

Impacts to estuaries (and other waterbodies) from nonpoint source pollution are wide ranging and come from every type of land use and many human activities (Table 1). Examples of impacts from nonpoint source pollution to estuaries include:

- Loss of species abundance
- Loss of species diversity
- Shellfish bed closures
- Swimming beach closures
- Decline in species health
- Fish kills
- Algae blooms (including toxic algae)
- Human disease outbreaks
- Flooding
- Low dissolved oxygen in water (NOAA, 2009)

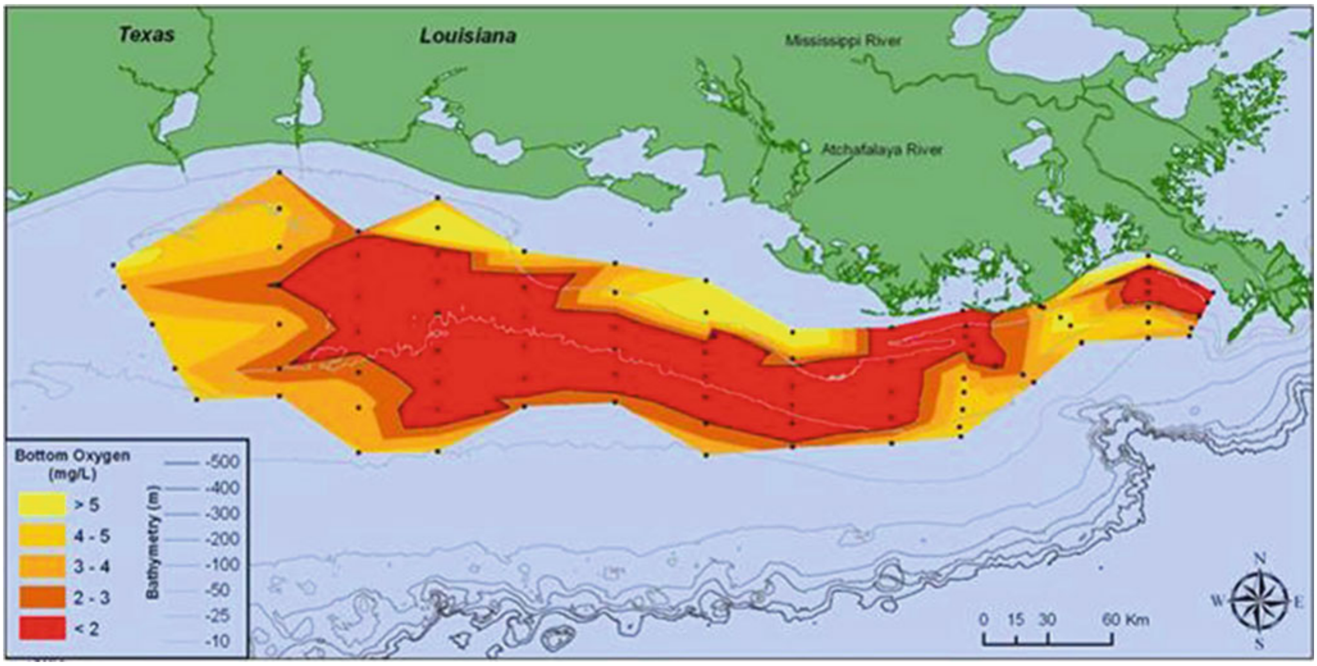
A study on eutrophication in estuaries, which is due in large part to nonpoint source pollution, reported that conditions in 65 % of the nation's estuaries will decline in the next decade (Bricker et al., 2007). A 1995 survey of shellfish waters in the USA found that 3.5 billion acres, or nearly one in every seven acres of shellfish beds, were unable to be harvested due to poor water quality. For over half of the coastal states included, nonpoint sources were responsible for more than 95 % of the areas closed to shellfishing (NOAA, 1996).

Major water quality issues across the country are caused by nonpoint source pollution in estuaries. Between 2004 and 2008, a hypoxic zone, characterized by dangerously low levels of dissolved oxygen in the northern Gulf of Mexico estuary, was measured to be, on average, 17,000 km^2 (the size of Lake Ontario; Figure 2). The reason for this catastrophe is the seasonal drop of dissolved oxygen levels in water to below 2 mg/L, where it is very difficult for living organisms to survive, and the source is the runoff from the Mississippi River Basin. Over time, there has been an increased use of fertilizers on agricultural land and combustion of fossil fuels to produce electricity and drive cars (LUMO, 2013). Although there are continued point sources of pollution that contribute to the hypoxic zone, a significant portion of the impact can be attributed to nonpoint source pollution, including some of the sources listed above. The result is a drastic increase in nitrogen and phosphorus that runs off into or is deposited through the atmosphere onto waterbodies and makes its way into the northern Gulf (Bierman et al., 2008).

Sedimentation and increased turbidity is another very common impact of nonpoint source pollution. Sedimentation rates in estuaries are naturally high, but human activities have significantly exacerbated the natural rates of filling through a variety of practices including poor soil

Nonpoint Source Pollution, Table 1 Activities that cause nonpoint source pollution by land use (Source: NOAA, 2009)

Forestry	Agriculture	Urbanization	Marina activities	Hydromodification
Road construction and maintenance	Concentrated animal feeding lots	Land clearing	Marina construction	Stream bank modification
Timber harvesting	Soil tillage	Road and building construction	Boat cleaning and painting	Stream channel modification
Pesticide and fertilizer application	Grazing activities	Fertilizer and pesticide application	Fueling	Dam construction and maintenance
Fire management	Fertilizer and pesticide application	Road salt application	Fish cleaning	Wetland loss and degradation
	Irrigation	Destruction of natural vegetation	Liquid and solid waste disposal	
		Wetland and riparian area destruction		
		Septic tank use		
		Pet wastes		



Nonpoint Source Pollution, Figure 2 Bottom-water dissolved oxygen concentrations in the northern Gulf of Mexico in late July 2013. Levels below 2 mg/L (shown in *red*) provide limited support to aquatic life (Source: LUMO, 2013).

conservation on agricultural land, altered circulation patterns, construction of dams and bridges, deforestation, and urban development. All of these activities accelerate coastal erosion and the suspended sediment concentration in estuaries (Schubel, 1977). A study conducted in the Albemarle-Pamlico estuary (NC) showed that nonpoint source pollution was the cause of over 96 % of stream degradation in the state, and agriculture was the source of 67 % of that total (Lilly, 1996). In the Tar Basin, 75 % of sediment load originates from cropland runoff. The erosion is due in part to the lack of investment in erosion control methods since leases on land are short term and investment in erosion control is not cost-effective at that time horizon (Lilly, 1996).

Regulations

The Clean Water Act (CWA) is the major federal law that is designed to protect the nation's waterbodies from pollution. While the CWA provides direct regulatory control over point sources through the NPDES program, there is no regulatory control over nonpoint sources. The 1987 amendments to the CWA established the Section 319 Nonpoint Source Management Program. It recognizes that without acting to control nonpoint source pollution, waterbodies in the USA will not meet water quality standards. Thus, the program currently provides grant funding to states to develop nonpoint source management plans or maintain programs that address this type of pollution (USEPA, 2013).

Coastal states are regulated by the Coastal Zone Act as well. Section 6217 of the Coastal Zone Act Reauthorization Amendments of 1990 (CZARA) requires coastal states (including Great Lakes states) to develop coastal zone nonpoint source management programs to be approved by both the USEPA and the National Oceanic and Atmospheric Administration (NOAA).

Management options

In addition to providing funding, the USEPA is also required by CZARA to develop a set of management measures that can be implemented to help reduce nonpoint sources of pollution for coastal states. The management measures are organized by type of source/land use and summarized in Table 2.

Stormwater from urban areas in all states are regulated by the NPDES program, which requires the development of stormwater management plans by qualifying cities, counties, and other urbanized areas. Each plan must include five major types of minimum control measures that reduce nonpoint source pollution. These control measures, and examples, are summarized in Table 3. The bag law in Washington, D.C. (a city which drains to the Potomac Estuary and then Chesapeake Bay), is an example of a local policy to prevent nonpoint source pollution of trash. This law enforces a five cent fee for the use of any plastic bag at all businesses that sell food or alcohol throughout the city. It has resulted in a significant reduction in trash, and money generated from the bag fee goes to fund trash cleanups and other programs to improve water quality in D.C.

Nonpoint Source Pollution, Table 2 Coastal management measures for the control of nonpoint source pollution (Source: USEPA, 1993)

Land use/source	Types of management measures	Target pollutants
Agricultural	Sediment/erosion control	Nutrients
	Confined animal facility operational changes	Pesticides
	Nutrient management	Sediment
	Pesticide management	
	Livestock grazing management	
	Irrigation management	
Urban/stormwater	Construction erosion/sediment control	Sediment
	Construction site chemical control	Bacteria
	Operation of on-site disposal systems	Toxics
	Siting roads, highways, and bridges	
Forestry	Preharvest planning	Sediment
	Streamside management areas	Nutrients
	Road construction/reconstruction	
	Road management	
	Timber harvesting	
	Site preparation and forest regeneration	
	Fire management	
	Revegetation of disturbed areas	
	Forest chemical management	
	Wetland forest management	
Marinas	Marina flushing	Metals
	Shoreline stabilization	Bacteria
	Fueling station design	Oil and grease
	Sewage facilities	
	Fish waste management	
	Boat cleaning	
	Petroleum control	
Hydromodification	Stream restoration	Sediment
	Erosion and sediment control	Toxics
	Chemical and pollutant control near dams	
	Evaluation of siting, construction practices, and operation of dams	
Wetland	Protection	Nutrients
	Restoration	Bacteria
	Vegetated treatment systems	Sediment

Summary

Nonpoint source pollution is a major issue preventing thousands of US waterbodies, including most estuaries, from meeting water quality standards. The impacts are pervasive and harmful to humans, wildlife, and the environment. The sources and pathways are diverse and difficult to define, which makes identification and management complex. Federal regulations enforced by the USEPA do not currently provide direct regulatory control of nonpoint sources of pollution, but there is funding and resources available to plan for and manage their impacts. NOAA's recent report on conditions in estuaries calls for "coordinated and integrated action that balances management action, efficient monitoring to assess the effectiveness of the management, targeted research, and a communications campaign aimed at engaging the broader community" (Bricker et al., 2007). Thus a multipronged approach is needed in order to address this challenging issue.

Nonpoint Source Pollution, Table 3 Minimum control measures for stormwater management plans

Minimum control measure	Examples
Public education/outreach	Fact sheets Webcasts Storm drain stenciling Recreational guides Educational programs for school-age children Stormwater hotlines Signage
Illicit discharge detection and elimination	Oil and hazardous chemical recycling Mapping stormwater outfalls Monitoring dry weather discharge Field screening
Construction site runoff control	Site plan review Inspections Silt fences
Post-construction runoff control	Buffer strips Minimization of impervious surfaces Open space requirements Stormwater detention and retention structures Porous pavement Rain gardens

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Cross-references

Anoxia, Hypoxia, and Dead Zones
Clean Water Act
Coastal Erosion Control
Eutrophication
Nitrogen
Phosphorus
Water Quality
Watershed

NONSTATIONARY FORCING

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Definition

The met-ocean forcing agents relevant for estuarine dynamics (e.g., sea-level pressure, wind, rain, water levels) have several temporal variability scales. The analysis of these variables is divided into short-term and medium- and long-term analysis. Medium- and long-term analysis is focused on understanding and modeling time series of state variables (e.g., mean wind speed at 10 m height, significant wave height) whose characteristic time scale is O (~hours). On the other hand, short-term analysis focuses on the instantaneous variables, whose characteristic time scale is O (minutes to seconds).

The processes whose characteristic time scale is O (days to weeks), such as synoptic scale processes and cycles of spring and neap tides, lead to autocorrelation in the time series of the state variables, as well as to cross-correlation among variables generated by the same physical process. An example is the passage of a storm: strong winds are generated, leading to local waves and storm surge. Then, the processes of time scale O (days to weeks) are superimposed on the processes of scale O (months) (e.g., seasons), which in turn are modulated by pluriannual processes O (years) such as El Niño/Southern Oscillation, as well as by long-term trends and processes of larger time scales (e.g., climate change).

For analyzing and modeling, the met-ocean agents in the medium- and long-term, multivariate nonstationary stochastic models, with multiple scales of variation, are required. Monbet et al. (2007) review several models applied for modeling and simulation of wind and wave time series. The methods are classified as parametric and nonparametric. The Translated Gaussian Process (TGP) method (Walton and Borgman, 1990) is the most widely used nonparametric method, while the most frequently used parametric methods are based on autoregressive models (e.g., Guedes Soares et al., 1996; Stefanakos et al., 2006). More recently, Solari and Losada (2011) introduced a parametric methodology based on the use of nonstationary mixture distribution functions and copulas.

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Cross-references

Estuarine Circulation
Tidal and Nontidal Oscillations
Tidal Hydrodynamics

NUTRIENT DYNAMICS

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Definition

Nutrient dynamics is broadly defined as the way nutrients are taken up, retained, transferred, and cycled over time and distance, in an ecosystem (Hauer and Lamberti, 2006; Allan and Castillo, 2007).

Description

Nutrients are defined in the broad sense as all chemical elements vital to biological functions (e.g., survival and reproduction) (DeAngelis et al., 1989). Organisms require macronutrients in large quantities such as carbon (C), hydrogen (H), nitrogen (N), phosphorus (P), potassium (K), sulfur (S), and calcium (Ca); other nutrients such as silicone (Si) are used for critical cellular processes. Micronutrients, such as copper (Cu) and zinc (Zn), are needed in lesser quantities (Smith and Smith, 1998). In nature, various elements can limit primary productivity of an ecosystem at a given place and time. The productivity of most aquatic ecosystems is controlled by the concentration, molecular form, and stoichiometry of the macronutrients N and P (Officer and Ryther, 1980).

Nutrient limitation

The German agricultural chemist, Justus von Liebig theorized that the yield of plants can be limited by the nutrient that is present in the environment in the least quantity relative to plant demands for growth (von Liebig, 1855); this theory has come to be known as Liebig's Law of the Minimum. The concept is the cornerstone of nutrient dynamics. It implies (1) that one key nutrient should be the primary limiting factor for plant growth in a given ecosystem, (2) that the growth of plants in a given ecosystem should be proportional to the rate of supply of this nutrient, and (3) that the control of eutrophication should be accomplished by restricting the loading of this key nutrient to the ecosystem (Smith et al., 1999).

Nutrient uptake

Nutrients are taken up by plant organisms and are affected by a number of abiotic (nonliving) and biotic (living) environmental interactions. Conditions that affect nutrient uptake in soil include soil texture, soil organic matter, soil water content, and soil temperature. Nutrient uptake is a function of plants and soils and is influenced by two main processes: (1) mass flow and (2) diffusion. Mass flow in soils is a rapid process, whereas diffusion is only measured in mm per day. Mass flow is insufficient to satisfy plant demand, but zones of nutrient depletion create concentration gradients that drive nutrient diffusional process in the soil.

Nutrient cycling

Ecosystems that receive chronically low inputs of limiting nutrients are characterized by mechanisms and structures that result in a high degree of nutrient recycling in relation to the amount of nutrient input. If there is no loss of nutrients to the ecosystem, the cycle is said to be a "perfect cycle," and if loss does occur, the cycle is said to be "imperfect." The decomposers play an important role in these cycles because they break down dead organisms and make the nutrient components available once more to other organisms. Microorganisms play an important role in the cycling of nutrients because they assimilate (take up) and mineralize nutrients and return them back to the nutrient pool.

For example Denitrification is an anaerobic microbial process that reduces nitrogen oxides (NO_3^-) to nitrogenous gases (N_2).

Controlling factors

The primary abiotic factors that control nutrient dynamics in ecosystems at a larger scale include the atmosphere, climate and water, soil structure and chemistry, water chemistry, and seasonality. Hydrologic processes control how much (amount or load), when (storm events), and where (estuaries) nutrients are deposited on the land and seascape. The hydrology, ecology, and biogeochemical processing are strongly coupled. In river basins, soils, groundwater, riparian zones and floodplains, streams, rivers, lakes, and reservoirs act as successive filters to retain a significant fraction of the nutrients transported (Bouwman et al., 2013). At smaller scales, components of abiotic factors, such as pH, salinity, redox conditions, nutrient availability, dissolved oxygen, and temperature, also play an important role in regulating the transfer, retention, and cycling of nutrients in an ecosystem. Biotic components such as microorganisms, plants, and animals take up (assimilate) and mineralize, retain, and cycle nutrients. Their role as producers, consumers, detritivores, decomposers, parasite, host, predator, competitor, herbivore, symbiant, and pathogens is an integral part in how nutrients flow through an ecosystem.

Nutrients sources

The sources of nutrient inputs are important to how elements are used and cycled in a biological system. Soil is composed primarily of weathered materials, along with water, oxygen, and organic materials. Most elements are released on land and transported to the ocean where they are eventually buried in marine sediments. Prior to reaching the ocean, nutrients are filtered through soils, groundwater, riparian zones, floodplains, rivers, lakes, estuaries, and coastal marine areas. Many of the nutrients are retained in these systems. Changes in land use and human activities have dramatically increased nutrients in terrestrial and aquatic ecosystems from sources such as fertilizer, animal manure, atmospheric deposition of nutrients, and wastewater flows (Stumm, 1973; Valiela et al., 1992;

Galloway et al., 1995; Vitousek et al., 1997). Increased nutrient loading, in particular N and P, to freshwater, estuarine, and marine ecosystems can lead to eutrophic impacts manifested by increased plant growth and in most cases undesirable changes in ecosystem structure and function. Large-scale human activities such as deforestation and expanding agricultural land use have caused increased erosion and sediment, C, and nutrient transport through riverine systems. Urbanization plays a major role in nutrient source contribution, and most impacts can be ascribed to a few major large-scale impacts, such as urban storm water runoff, combined or sanitary sewer systems, waste water treatment, and legacy pollution (Walsh et al., 2005). These increases in nutrients from various sources such as urban landscapes, coastal communities, and agricultural and suburban environments impact the transport and retention of nutrients, most notably N and P, in the environment (Grimm et al., 2005; Collins et al., 2010; Harrison et al., 2011).

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Cross-references

Macronutrients
Micronutrients
Nitrogen
Nutrient Limitation
Nutrients
Phosphorus

NUTRIENT LIMITATION

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Synonyms

Nutrient deficiency; Nutrient insufficiency

Definition

Nutrient limitation is defined as growth limitation of an organism due to nutrient deficiency. A nutrient that is in lowest supply relative to the others will be the limiting nutrient to growth and productivity of the organism (Liebig's Law of the Minimum).

Description

Ecological Importance: In aquatic ecology, the term "nutrient limitation" refers to the limitation of net primary production, or total system net production. The concept of a limiting nutrient is essential to understanding biological processes. The nutrient in short supply relative to the others will be exhausted first and will thus limit cellular growth. Nutrient limitation leads to a reduction in growth and photosynthetic rate (Falkowski and Raven, 2007). However, unless the nutrients are available in adequate amounts relative to each other, autotrophic growth is "nutrient limited" by one or the other nutrient. Carbon, a highly available element in the estuarine environment (carbonate, bicarbonate, and carbon dioxide), is rarely limiting. Minor and trace elements are required by autotrophs, although their estuarine availability usually is not limiting to growth (Day et al., 2012). Nutrient elements are transferred between organisms and their environment, and one organism's wastes can become another organism's nutrients.

Nutrients Ratio: Nutrient limitation indicates that two nutrients are out of the appropriate balance. The optimal nitrogen/phosphorus ratio (N/P-ratio) for phytoplankton growth is 16:1 (based on molar concentrations) and is called the Redfield ratio (Redfield et al., 1963). Phosphorus limitation occurs when there is proportionally less phosphorus than nitrogen (i.e., there is excess nitrogen). Significant deviations from 16 at low N/P-ratios might indicate potential nitrogen limitation and, at high N/P-ratios, potential phosphorus limitation of phytoplankton primary production. Phosphorus limitation occurs in the spring in some estuaries when nitrogen availability is high from storm water runoff. Nitrogen limitation occurs when there is proportionally less nitrogen than phosphorus (i.e., there is excess phosphorus). This might affect the biological state of the ecosystem, in particular the phytoplankton biomass, species composition, and eventually food web cycles.

Human Interferences: N:P ratios in estuaries have commonly been shown to be lower and/or higher than the predicted Redfield ratio because of denitrification and anthropogenic nutrient enrichment processes, respectively (Day et al., 2012).

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Cross-references

Bioavailability
Ecological Monitoring
Ecological Stoichiometry
Macronutrients
Micronutrients

NUTRIENTS

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Definition

Nutrients are elements that are functionally involved in the key processes of life. They are chemical elements or compounds required for autotrophic growth and metabolic function of estuarine organisms.

Ecological importance

Nutrients are chemical elements of great ecological importance. They include essential elements necessary for primary producers and for metabolic activities of primary consumers (including C, N, P, Si, S, K, Mg, Na, Fe, Mn, Zn, Cu, B, Mo, Co, V, and vitamins). Different organisms have specific elemental requirements. For example, aquatic plants have the greatest demand for carbon, nitrogen, and phosphorus. Silicon is an element heavily utilized by diatoms (Day et al., 2012).

Cycles

The chemical constituents of estuarine organisms are continually cycled among the organisms, water, sediments, and atmosphere. Nutrient cycling in estuaries is greatly influenced by the microbial community. Nutrient elements have cycles that alter their availability in organic and inorganic forms while maintaining continuous production in estuarine systems (Bianchi, 2007).

Macronutrients

Macronutrients are those nutrients most greatly assimilated by algae and vascular plants in estuaries. Nitrogen (N), phosphorus (P), and silicon (Si) are limiting because carbon (C) is available in large quantities, and the biotic demand for trace elements is low. Carbon is found in all aquatic ecosystems and is rarely limiting. Organic matter in estuaries primarily consists of carbon, hydrogen, oxygen, nitrogen, sulfur, and phosphorus. Phytoplankton and other primary producers assimilate dissolved inorganic nutrients and transform them into organic compounds.

Carbon

Carbon is readily available in the marine environment in carbonate, bicarbonate, and carbon dioxide which are components of the carbon cycle. Organic carbon is added to an estuary in both dissolved and particulate forms (i.e., autochthonous and allochthonous sources). Allochthonous input can occur from marine and terrestrial sources, as well as atmospheric transport. Human impact on the carbon cycle is important. Increased concentrations of carbon dioxide, methane, and nitrous oxide in the atmosphere are linked with global warming through the greenhouse effect (Stumm and Morgan, 1996). Carbon dioxide water-to-air fluxes are significant in estuaries (Bianchi, 2007).

Nitrogen

The atmosphere is the largest reservoir of nitrogen which primarily occurs in diatomic molecular form (N_2). Quantitatively, the most important forms are total ammonia nitrogen ($NH_3 + NH_4^+$), nitrate (NO_3^-), and nitrite (NO_2^-). Nitrogen limitation leads to a reduction in autotrophic growth and photosynthetic rates (Falkowski and Raven, 2007).

Phosphorus

Phosphorus is available in estuaries as dissolved inorganic phosphorus (DIP), dissolved organic phosphorus (DOP), and particulate organic phosphorus (POP). DIP is rapidly assimilated by algae during the photosynthetic process. It is essential for autotrophic growth and development in estuaries.

Silicon

Silicon (Si) is a chemical element with atomic number 14 and atomic weight 28.0855. Silicon dioxide (SiO_2) occurs as silicate minerals in the earth's crust. It is found in natural waters as dissolved reactive silicon and orthosilicic acid ($Si(OH)_4$) or $H_4\,SiO_4$). Diatoms, radiolarians, and siliceous sponges use biogenic silica as structural material to construct skeletons. Silicon is assimilated by diatoms at rates similar to that of nitrogen, and it enters estuaries via river flow carrying soil-leached products. The crystalline form of silicon in skeletal material is "biogenic silica." The Si:N ratio influences the composition of phytoplankton. The growth of diatoms depends on the presence of silicate, while the growth of non-diatomaceous forms normally does not (Day et al., 2012).

Sulfur

Sulfur (S) is a chemical element with atomic number 16 and atomic weight 32.065. It is an essential element important for aquatic biochemical processes, metal cycling, and ecosystem energetics (Day et al., 2012). It occurs naturally as elemental sulfur, sulfide, and sulfate minerals. In metabolic reactions, sulfur compounds serve as both fuels (electron donors) and respiratory (oxygen alternative) materials (electron acceptors). In anaerobic sediments and pore water of estuaries, sulfate reduction is an important process that passes energy from autotrophs through the food web. Coal, biomass burning, and volcano emissions inject SO_4 into the atmosphere, which can then be further oxidized in the atmosphere and removed as ${SO_4}^{2-}$ in precipitation (Bianchi, 2007). Sulfur is widely used in agriculture as a fertilizer which can be an important anthropogenic source of the element to receiving waters.

Estuarine behavior

Nutrients are affected by estuarine mixing and circulation processes. They are typically present at the air-water, river-sea, land-sea (salt marshes), and sediment-water interfaces. Nutrient concentrations in an estuary are influenced by their hydrodynamic characteristics. Nutrient distributions in an estuary are determined in part by the basin morphology, circulation, and tidal regime; biological processes and regeneration; organic detritus accumulation; and transport. Gradients in estuarine processes are primarily driven by changes in river inflow, tidal currents, waves, and meteorological forcing (Bianchi, 2007; Day et al., 2012). Anthropogenic activities can alter the balance of estuarine nutrients.

Micronutrients

Micronutrients are trace elements essential for autotrophic organisms in small amounts. They are necessary for photosynthesis, being key constituents of chlorophyllous pigments and enzymes (Falkowski and Raven, 2007). The trace elements include iron, cobalt, chromium, copper [2], iodine, manganese, selenium, zinc, and molybdenum. Iron has been reported by many researchers as the most important bioactive trace element (Day et al., 2012).

Sources

As noted above, nutrient sources may be autochthonous, if they result from organic matter remineralization in the estuary, or allochthonous, if they enter the estuary from external sources such as river input or atmospheric deposition. Free nitrogen (N_2) fixation by planktonic organisms in estuaries is typically low, even when the N:P ratio is low. The trend toward lower fixation may be due to low availability of one or more trace elements required for N fixation, including molybdenum and iron. Estuaries often become highly productive areas due to enrichment through soil weathering and nutrients supplied by rivers which can support dense phytoplankton populations.

Limitation

Nutrient limitation leads to a reduction in autotrophic growth and photosynthetic rates. According to Liebig's law of the minimum, the concentration of a nutrient at the smallest relative amount required for organism growth may be the limiting factor. The concept of a limiting nutrient is essential to understanding biological processes. The nutrient in short supply relative to the others will be exhausted first and will thus limit cellular growth (Falkowski and Raven, 2007). Physical processes (e.g., transport, sedimentation/resuspension, and sorption/desorption) together with biological processes (e.g., regeneration and bioturbation) can significantly alter nutrient concentrations in estuaries, whereby they exhibit non-conservative behavior. The nutritional requirements of phytoplankton have a relative ratio of nutrients, which may be related to the relative proportion of the elements available in the estuarine environment. Nitrogen is the limiting nutrient in most estuarine waters because it is in higher demand and is depleted more rapidly compared with other nutrients. It also has a more complex biogeochemical cycle than phosphorus (Day et al., 2012).

Eutrophication

Nutrients are essential to the trophic web and estuarine biodiversity, although excessive nutrient enrichment results in eutrophication and environmental imbalance. Human activities have increased the input of nitrogen and phosphorus to estuarine and coastal waters, thereby increasing the Si:N and Si:P nutrient ratios and causing the loss of estuarine biodiversity (Bianchi, 2007). Increased anthropogenic N flow into rivers from agricultural, industrial, and urban activities intensifies the estuarine eutrophication process to favor the growth of algae and other unwanted vegetation (Kennish, 1997). Eutrophication can cause major changes in phytoplankton community structure, the proliferation of harmful or toxic algae, decline of fisheries, and loss of biodiversity (Stumm and Morgan, 1996; Kennish, 1997).

Summary

Nutrients are chemical substances of great ecological importance, being essential for primary production in estuaries. They are essential to the trophic web and estuarine biodiversity. Iron is the most important bioactive trace element. Nutrient elements are continually cycled among organisms, water, sediments, and the atmosphere. The concentrations of nutrients in estuaries are affected by mixing and circulation processes. Nutrient enrichment can lead to eutrophication and environmental damage, compromising estuarine community structure and function.

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Cross-references

Ecological Stoichiometry
Ecological Monitoring
Eutrophication
Macronutrients
Micronutrients
Nitrogen
Nutrient Dynamics
Nutrient Limitation
Phosphorus

O

OIL POLLUTION

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Synonyms

Crude oil; Petroleum

Definition

Crude oils consist of complex mixtures of hydrocarbon and nonhydrocarbon compounds, and they vary considerably in their chemical composition and physical properties. Discharges, leakages, or spills of oil can cause serious impacts in estuarine and marine environments.

Composition

Hydrocarbon compounds account for more than 75 % of most crude oils, although nonhydrocarbon components (e.g., compounds containing oxygen, nitrogen, sulfur, and metals such as copper, iron, nickel, and vanadium) can predominate in heavy crude oils. There are many toxic substances in crude oils including, but not limited to, benzene, toluene, xylene, phenols, and carboxylic acids. Hence, oil spills and leakages into estuarine and marine environments can pose a serious threat to biotic communities and habitats in these environments, particularly those in the coastal zone (Kennish, 2001; Cormack, 2010).

Crude oils are composed of four major classes of hydrocarbons: (1) straight-chain alkanes (*n*-alkanes or *n*-paraffins), (2) branched alkanes (isoparaffins), (3) cycloalkanes (cycloparaffins), and (4) aromatics. The lower-molecular-weight compounds in each class dominate in these oils (Kennish, 1997). Several important physical-chemical processes alter the composition and toxicity of oil in estuarine and marine environments through time. These include evaporation, photochemical oxidation, emulsification, and dissolution. Marked changes in composition occur during the first 24–48 h of an oil spill when the more toxic and volatile components are lost to evaporation and dissolution effects.

Anthropogenic effects

Most oil inputs from human activities to estuarine and marine environments result from marine transportation (deballast and tanker accidents), leakages of fixed installations (e.g., coastal refineries, offshore production facilities, and marine terminals), municipal and industrial wastewaters, urban and suburban runoff, and atmospheric deposition (Kennish, 1997; Kennish, 2001). A number of factors determine the severity of oil impacts on organisms and habitats in these environments: (1) amount of the oil; (2) composition of the oil; (3) form of the oil (i.e., fresh, weathered, or emulsified); (4) occurrence of the oil (i.e., in solution, suspension, dispersion, or adsorbed onto particulate matter); (5) duration of exposure; (6) involvement of neuston, plankton, nekton, or benthos; (7) juvenile or adult biotic forms exposed; (8) previous history of pollutant exposure; (9) season of the year; (10) temperature, salinity, and other physical-chemical variables; (11) type of habitats impacted; and (12) cleanup methods (Clark, 1992; Doerffer, 1992; Kennish, 1992; Kennish, 2001). Polluting oil may not only be directly toxic to aquatic organisms, but also can increase their mortality by smothering or suffocating them. Both lethal and sublethal effects can devastate impacted populations. Therefore, oil spills and other releases of oil to estuarine and marine environments must be carefully monitored and quickly remediated (Kim et al., 2010).

Oil spills tend to sink through time in the marine hydrosphere as the density of the oil increases. Hence, sedimentation of oil on the seafloor poses a significant long-term

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threat to benthic communities and the food web. Once the oil enters the benthic regime, bottom sediments, and wetlands habitats, it is extremely difficult to remove and prevent ecosystem damage. It is not unusual for heavy oil pollution to impact benthic habitats and organisms for a decade or more. Microbes (bacteria, in particular, and fungi) are the primary biotic agents responsible for degrading oil in aquatic environments (Das and Chandran, 2011; Kostka et al., 2011). The rate of biodegradation of polluting oil depends greatly on the water temperature, nutrient availability, oxygen levels, and salinity of the impacted water body.

Conclusions

Oil pollution is both a potentially acute and insidious problem in estuarine waters around the world. While there is great fear of major oil spills that directly impact these waters, chronic oil pollution can cause serious insidious adverse effects on biotic communities and habitats. Estuarine benthic, littoral, and wetlands communities and habitats are especially vulnerable and can be altered for decades by oil pollution.

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Cross-references

Anthropogenic Impacts
Ecological Monitoring
Microbial Degradation
Nonpoint Source Pollution
Oil Pollution
Polycyclic Aromatic Hydrocarbons

OVERWASH

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Definition

Overwash is the flow of water and sediment over the crest of a beach system when the run-up level of waves or the water level, often enhanced by storm surge, exceeds the local beach or dune crest height (Donnelly, 2008).

Description

Overwash commonly results from hurricane storm surges, although any storm that raises water levels sufficiently to overtop the berm crest or dune crest of a beach system can cause overwash. Depending on the relative height of water levels and the beach system, two levels of overwash may be defined: (1) overwash regime, where water levels are sufficient to overtop low spots in the berm crest or dune crest, and (2) inundation regime, where water levels completely overtop the beach and/or dunes and the entire beach system becomes subaqueous (Sallenger, 2000). Overwash can significantly impact the geomorphology of beach systems by eroding the nearshore zone, beach, and dunes and by creating sediment deposits – known as washover – landward of the beach system. Under overwash regime conditions, the flow of water and sediment is funneled through low spots in the berm or dune crest and deposits isolated, lobate-shaped washover fans in the backshore area, typically extending tens to hundreds of meters inland. If overwash occurs along a longer section of a beach system, washover deposits coalesce into a continuous sediment apron known as a washover terrace. Under inundation regime conditions, washover sediments may extend several kilometers inland, forming large sheetlike deposits in the backshore area. Because overwash of a barrier island causes a net onshore transfer of sediment, it may contribute to the gradual landward migration of the island – a process known as “rollover” (Orford and Carter, 1982).

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Cross-references

Washovers
Washover Fans

OXYGEN DEPLETION

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Synonyms

Deoxygenation

Definition

Oxygen depletion in the aquatic environment is a process involving the decrease in the concentration of dissolved oxygen caused by its consumption during the oxidation of organic matter and/or reduced compounds of other chemical elements. Oxygen depletion leads to changes in *redox conditions*.

Description

There is a normal sequence or succession of processes during the transition from oxic to anoxic conditions. The bacterial decomposition of organic matter is an oxidative process. If there is an excess of organic matter to be decomposed once the dissolved oxygen has been used, bacterial activity will substitute the dissolved oxygen with a series of other electron acceptors (i.e., nitrate through *denitrification*) and metals that usually end with the reduction of sulfate, a major constituent in seawater. This last process produces hydrogen sulfide, which is toxic to aerobic life forms. Conversely, the oxidation of reduced inorganic compounds under anoxic conditions also fuels the microbial production of organic matter via chemosynthesis (Nealson and Stahl, 1997; Sorokin, 2002; Canfield et al., 2009).

The occurrence of oxygen-depleted and anoxic water depends on the combined influence of eutrophication (organic matter and nutrient loads) and hydrodynamics (intensity of mixing and water renewal) (Yakushev and Newton, 2013). Oxygen depletion zones form when there is an imbalance between the supply of organic matter and the supply of dissolved oxygen for its decomposition. This may occur when a hydrophysical structure, such as temperature stratification, restricts the aeration of the water column. The existence of low oxygen structures can be temporary or permanent, correspondingly creating zones of temporary or permanent hypoxia and anoxia.

Oxygen depletion and anoxia in the water column are increasingly common features observed in the world ocean, inland seas, estuaries, and coastal marine waters. Observations show a decline in the dissolved oxygen concentrations at the continental margins in many regions, and these are related to both an increase in anthropogenic nutrient loading and a decrease in vertical mixing (e.g., Richardson and Jorgensen, 1996; Diaz, 2001; Rabalais et al., 2002; Diaz and Rosenberg, 2008; Savchuk, 2013). Decreases in oxygen are also reported in the tropical oceans (Emerson et al., 2004; Whitney et al., 2007; Paulmier and Ruiz-Pino, 2009; Keeling et al., 2010; Deutsch et al. 2011; Falkowski et al., 2011). Low oxygen "dead zones" have spread exponentially since the 1960s (Diaz and Rosenberg, 2008). The decrease in dissolved oxygen throughout the open ocean basins may be a long-term, nonperiodic trend related to climate change, or the result of natural cyclical processes, or a combination of both (Falkowski et al., 2011). In the coastal regions, oxygen depletion events are connected with anthropogenic forces such as eutrophication. However, climatic forces may further stimulate the formation of oxygen-depleted zones. Within the marine science community, there is increasing interest in these events because of their global occurrence.

The scale of processes that affect the formation of each system's hydrophysical structure varies from molecular diffusion to climatic variability. However, in lakes and fjords, some processes, such as the transport of water with geostrophic currents or mesoscale eddies, are not important, leading to less intense mixing than under marine conditions. Therefore, the chemical structure of redox interfaces in fjords and lakes is characterized by sudden changes in redox conditions and steep chemical gradients. In comparison to lakes and fjords, oxidation-reduction features at marine redox interfaces are characterized by gradual gradients, with varying temporal changes as well. For example, the boundary of the anoxic zone of the Cariaco Basin is influenced by mesoscale eddies that periodically supply dense water with high oxygen content to the anoxic zone (Scranton et al., 2006). In the Baltic Sea, under certain winter weather conditions, there is an influx of oxygen-rich saline Northern Sea waters to the deep anoxic layers (Schneider et al., 2002; Feistel et al., 2008). The Black Sea oxic/anoxic interface appears to be more stable, because the Bosphorus plume waters influence only the southwestern part of the Black Sea. The central and peripheral Black Sea is characterized by the stability of the chemical features (i.e., maximum positions, onset levels) in the density field (Vinogradov and Nalbandov, 1990; Murray et al., 1995). Such a *chemotropicity* implies that there are no horizontal gradients of chemical variables along the same density surface, supporting the use of a "one-dimensional modelling" for describing the processes responsible for the maintenance of the redox interface chemical structure (Yakushev et al., 2005).

The redox interface is a layer where oxic and anoxic conditions are adjacent. It is very complex because of the many chemical reactions and biogeochemical mineralization processes that can be oxic, suboxic, and anoxic. The depletion of oxygen concentration affects the biogeochemical cycles of N, P, and S, as well as the carbonate system and trace metal equilibria. Water layers that are already depleted with respect to dissolved oxygen may be close to the threshold between suboxic and anoxic conditions and are the water bodies most vulnerable to the effects of the globally observed deoxygenation trend (Jost and Pollehne, 2013). It is necessary to better understand the physical processes leading to anoxia, the biogeochemical structure of the oxic/anoxic interfaces, and the ecological consequences of oxygen depletion in order to predict the possible effect of global changes on oxygen conditions.

Oxygen depletion significantly affects water quality and ecosystem function. The absence of oxygen impairs the oxic ecosystem both directly and indirectly. Increased occurrence and volumes of anoxic water threaten the functioning of healthy aerobic ecosystems and thereby have a direct impact on human welfare and ecosystem services. There can be an additional, indirect impact at the water-sediment interface, where redox equilibria control the rate of supply of phosphorus from the sediments and the release of hazardous substances (e.g., methylmercury).

The consequence of oxygen depletion of natural waters is an important element of water-quality legislation (e.g., in the Water Framework Directive; EC, 2000).

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Cross-references

Anoxia, Hypoxia, and Dead Zones
Redox Conditions

OYSTER REEF

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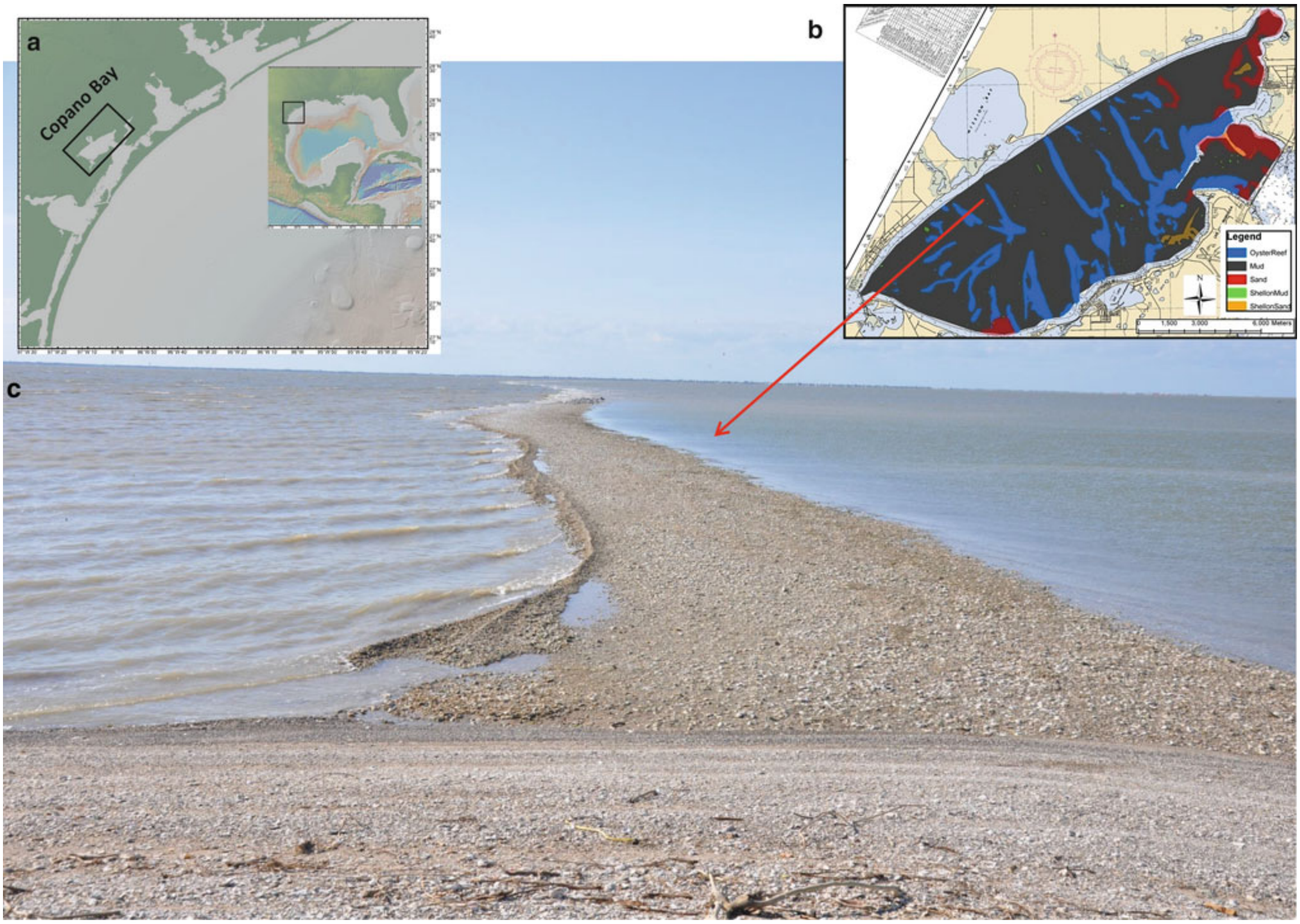
Definition

Oyster reefs consist of large clusters of oysters that form habitat in estuaries. Oysters play an important role in maintaining water quality and providing habitat for juvenile fish and other marine organisms (Kilgen and Dugas, 1989).

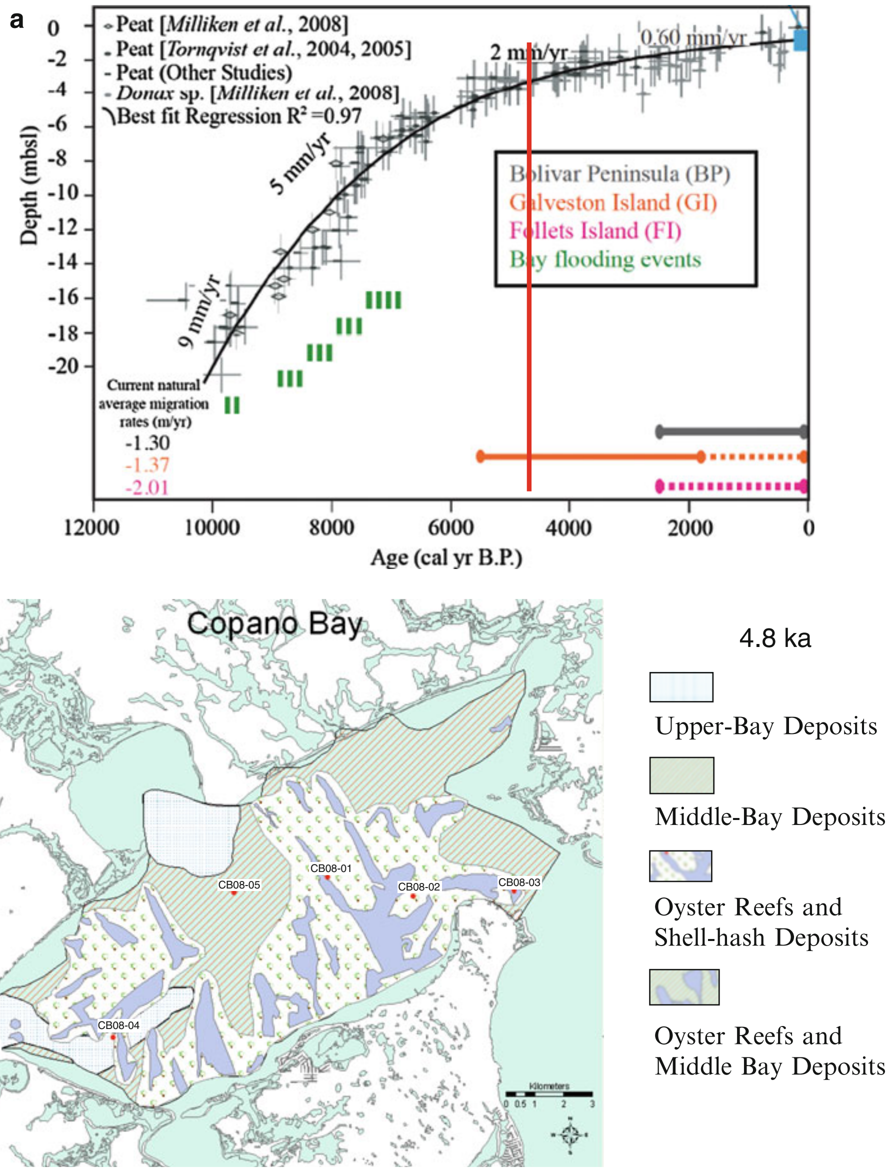
Description

Crassostrea virginica, also known as the American or Eastern Oyster, is the reef-building oyster species found within the estuaries and bays of the northern Gulf of Mexico and along the Atlantic coast of North America. *C. virginica* has a free-swimming larval stage, lasting 14–30 days, after which the larva goes through a settling or spatting process, in which it must permanently attach itself to the bottom (Kennedy et al., 1996). During the larval stage, the oyster is susceptible to the multitude of estuarine hydrodynamic processes that distribute the larva throughout the estuary. *C. virginica* is a filter feeder, so it needs to settle and attach to a site which will permanently keep it out of the fine bay bottom mud to avoid suffocation by fine-grained sediment as well as to optimize the supply of nutrients (Lenihan, 1999; Schulte et al., 2009). It also needs to settle within the intertidal to subtidal portion of the bay, to allow it access to tidal flow and nutrients (Kennedy et al., 1996).

For oyster reefs to form and persist, oysters need to accumulate on a stable substrate that keeps them elevated above the muddy bay bottom. Over time, for oyster reefs to survive, they need to form atop a substrate capable of supporting the weight of the reef. Coastal plain estuaries primarily form within partially filled incised river valleys (Pritchard, 1956). These river valleys have an inherited or "antecedent" geology resulting from the valley incision and valley fill. Studies on the geological controls on oyster reef distribution and occurrence within Copano Bay in Texas reveal that the major oyster reefs had been established by 4,800 ka (Figure 1), when sea-level rise rates had reduced (Piper, 2010; Troiani et al., 2011). Copano Bay resides within the partially filled incised valleys of the Copano, Mission, and Aransas rivers. These valleys have eroded through the Pleistocene-aged Beaumont Formation, consisting primarily of a hard, dense, and indurated clay with very high compressive strength. Seismic lines within the bay reveal that most of the large reefs are situated atop old river terraces, channel levees where the Beaumont Formation was not deeply incised (Figure 2), or atop tidal deltas and bayhead delta deposits where the Beaumont



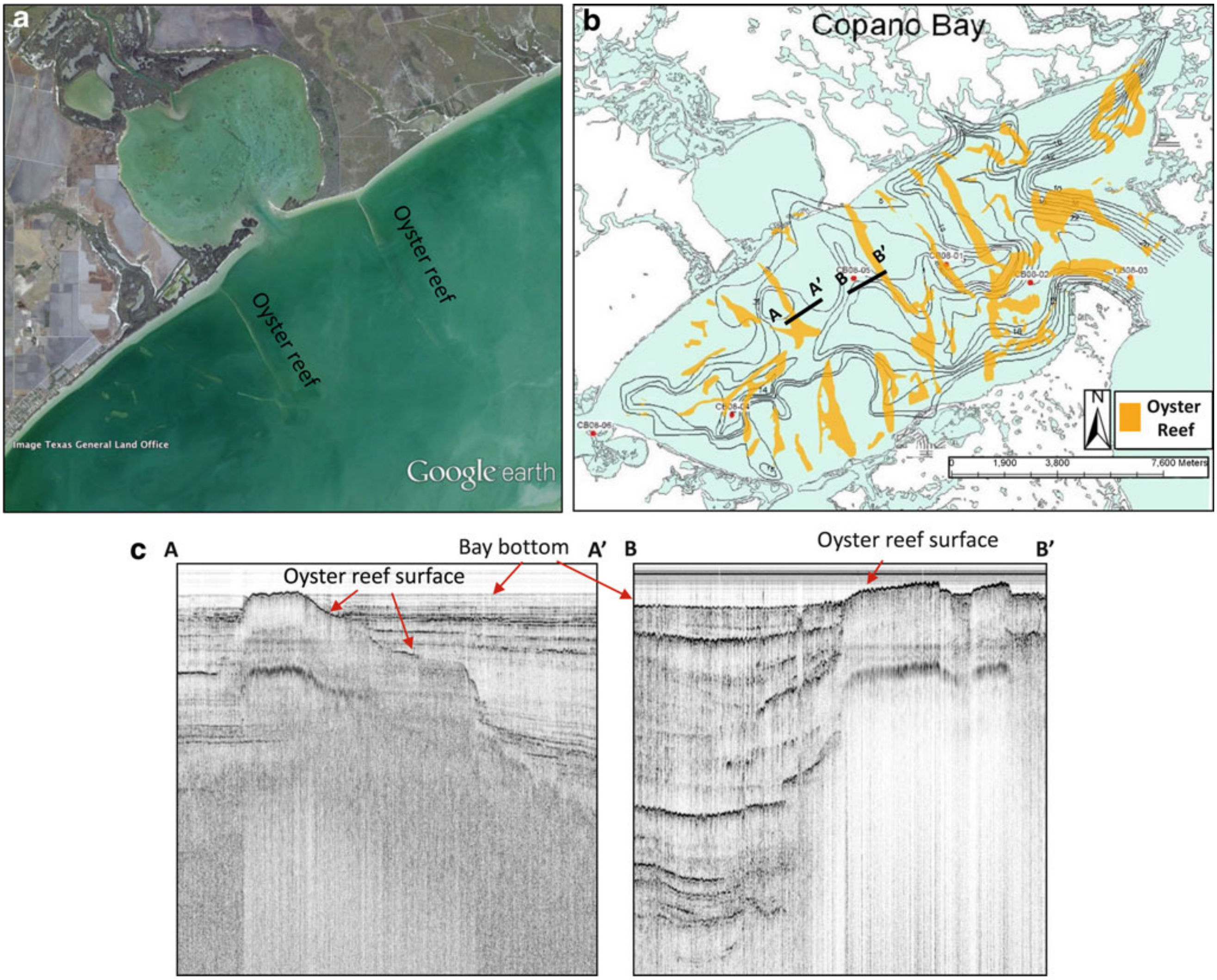
Oyster Reef, Figure 1 Oyster reefs in Copano Bay. (**a**) Inset map showing the location of Copano Bay along the northwestern shore of the Gulf of Mexico; (**b**) distribution of oyster reefs mapped in 2007 (Piper, 2010); (**c**) Google Earth image showing one of the large, shore-attached reefs that extends over 3.5 km into the bay.



Oyster Reef, Figure 2 Paleogeographic distribution of oyster reefs in Copano Bay from 4.8 ka. (**a**) Sea-level history of the Gulf of Mexico since the Last Glacial Maximum (Simms et al., 2007); (**b**) paleogeographic distribution of oyster reefs and oyster shell hash deposits 4.8 ka (Troiani et al., 2011), showing that 53 % of the Copano Bay bottom contained oyster reefs and shell scatter.

Formation was incised (Piper, 2010; Troiani et al., 2011). Within Copano Bay, 4,800 yBP, 53 % of the bay bottom was covered with oyster reef and oyster shell shoals (Figure 2); this constituted nearly the entire bay bottom located above the river terraces and levees. Surveys conducted in 2007 (Figure 3) revealed that, although there are very large oyster reefs within the bay, only 19 % of the bay bottom is reef and that these reefs are the peaks of much larger reefs that existed 4,800 yBP.

Antecedent geological controls on oyster reef distribution comparable to those found in Copano Bay have also been found in San Antonio Bay (Bouma, 1976), Lavaca Bay (Bronikowski, 2004), and Galveston Bay (Powell et al. 1995). In Apalachicola Bay, located along the panhandle of Florida, Twichell et al. (2010) found the oyster reefs formed around 5,100 Ka primarily atop buried delta deposits. Smith et al. (2003) found that buried river terraces provided the foundation to many of the oyster reefs found within the mesohaline portion of the Chesapeake Bay. As with Copano Bay, each of the other studies also revealed that the oyster reefs present today are the peaks of much larger reefs that existed in the geological past.



Oyster Reef, Figure 3 Oyster reef crests. (**a**) Google Earth image showing the crests of two prominent oyster reefs extending from the shoreline 3.5 km into the bay. The western reef is shown in Figure 1c. (**b**) The 2007 oyster reef distribution map showing the structural contours of the Pleistocene surface. (**c**) Two chirp seismic lines showing that the tops of the exposed reefs are part of much larger, older reefs that are now buried in bay mud.

In each case, antecedent geology, in the form of river terraces along the margins of incised fluvial valleys, tidal or bayhead deltas, or other large sand shoals provided a high-compressive-strength foundation for these reefs to form.

Because coastal plain estuaries exist within partially filled incised valleys, the portions of the bay bayward of river terraces, tidal deltas, and bayhead deltas tend to be areas where there are often 10–30 m-thick sequences of relatively soft, unconsolidated, bay fill muds. These areas are generally not capable of supporting oyster reefs. Oyster reef restoration projects often do not consider the geology when siting artificial reefs, and many restoration projects have failed because the shell sinks into deep bay fill deposits with no antecedent geologic foundation. Historical oyster reefs exist within the bay because there is a foundation for them. For oyster reef restoration projects to succeed, they need to site the reefs on areas where there is an antecedent geological foundation capable of supporting the reef.

Summary

Oysters need to accumulate on a stable substrate that keeps them elevated above the muddy bay bottom. Over time, for oyster reefs to survive, they need to form atop a substrate capable of supporting the weight of the reef. Oyster reef restoration projects often do not consider the geology when siting artificial reefs, and many restoration projects have failed because the shell sinks into deep bay fill deposits with no antecedent geologic foundation.

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Cross-references

Artificial Reef
Estuarine Habitat Restoration

P

PARTICIPATORY RESEARCH

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Synonyms

Action research; Collaborative research; Community-based participatory research; Transdisciplinary research

Definition

Participatory research is a methodological approach that emphasizes not only greater involvement of parties affected by the issue in question but also the sharing of power with those same participants (Cornwall and Jewkes, 1995). This is in contrast to the more traditional research approach wherein the power – within the research project itself – resides predominantly or wholly with the scientists and the project team.

Description

The espoused benefits of participatory research on estuaries have to do both with ethics and pragmatism. Ethically, participatory research can be seen as being more equitable, especially in situations where the issues being researched involve real-world societal problems and questions of justice, such as in the areas of community development, health policy, and environmental conservation.

Pragmatically, many argue that participatory research is more efficient and more liable to produce useful results. When those who are affected by the research are involved as colleagues in the research, helping with research design and implementation, the results are likely to be more relevant and trusted.

Since relevance is a key attribute in participatory research, it follows that most estuarine research conducted in this manner is what natural resource scientists refer to as "applied," as opposed to curiosity driven. In the health field, the corresponding terms are "knowledge for action" versus "knowledge for understanding."

Participatory research remains fairly controversial and misunderstood, both by scientists and nonscientists, many of whom consider it "soft" or less rigorous. However, there is nothing about participatory research that, by definition or actual practice, makes it softer or less rigorous. What is key about participatory process is the location of power. Since power tends to be shared by many, participatory research efforts are often complicated and challenging to implement.

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PATHOGENS

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Synonyms

Infectious agent

Definition

A pathogen is an infectious agent or, more commonly, a biological agent that causes disease or illness to its host. There are several substrates and

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pathways whereby pathogens can invade a host, and principal pathways have different episodic time frames.

Description

Estuaries form buffer zones between rivers and oceans, where many anthropogenic inputs and pathogens are trapped or modified via several mechanisms, including dispersal within the estuary, burial, wind-induced flushing, or storage in bottom sediments. The range of mechanisms operating in each estuary depends on hydrographic forcing, but the length of time pathogenic organisms remain viable and pose a risk to public health is contingent on their deposition, lifestyle in sediments, and resuspension. Further, pathogens have the potential to persist or even proliferate in estuaries (Noble and Fries, 2007). This suggests that, once fecal bacteria and other pathogens enter the aquatic environment, they should be able to establish long-term reservoirs, depending on their ability to acclimate and tolerate a range of environmental conditions (Miller et al., 2006).

Monitoring of microbial contamination of inland waters and estuaries usually focuses on their inputs during rainfall. Sediments may represent a significant source of pathogen populations during resuspension events that can be driven by wind without rainfall or by wind coincident with rainfall in shallow systems (Noble and Fries, 2007).

An increase of pathogen bacteria in invertebrates exposed to human sewage sources suggests that anthropogenic activities can have significant effects on the ecology of fecal bacteria pathogens at the land–sea interface. Additional research is needed to minimize the impacts of human and animal pathogens on estuarine and coastal marine environments for their long-term sustainability and health (Collins and Rutherford, 2004).

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Cross-references

Microbial Degradation
Microbial Survivability

PEAT

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Synonyms

Histosol; Humate; Peat soil; Turf

Definition

Peat is the accumulation of partly decomposed vegetable matter generally containing less than 20 % inorganic matter.

Introduction

Peat is an organic soil composed of partially to well-decomposed plant remains, which accumulate in a wet environment. Formation of peat occurs under wetland conditions, where land is saturated with water and flooding obstructs the flow of oxygen from the atmosphere which results in slowing down of decomposition rates (Keddy, 2010). Accumulation therefore occurs only under anoxic (i.e., no oxygen) conditions. Peat exhibits very unique physical and chemical properties unlike any other geological material, particularly in terms of porosity, water-holding capacity, and organic matter content. Peat soils are important sources of carbon-based greenhouse gases such as methane and carbon dioxide; however, the role of these gases in the global carbon cycle (particularly in terms of spatial and temporal variability in production and release to the atmosphere) is currently unclear. For that reason, the effect of peat-accumulating environments in future climate and global warming is currently not well understood. Although plant remains in current peats differ from those found in coals, peat is considered the precursor of coal after diagenetic and metamorphic compressional changes, and it is well accepted that most coal deposits of economic value formed from ombrotrophic (or precipitation-fed) peats rather than rheotrophic/minerotrophic (or groundwater-fed) peat (Clymo, 1987; Moore and Shearer, 2003).

Physicochemical characteristics of peat

Peat exhibits very unique physical properties when compared to inorganic sediments, including very low bulk density and a very high water-holding capacity. Bulk density (or dry weight of soil per unit volume) tends to range between 0.2 and 0.3 g cm^{-3} for well-decomposed peat; however, certain peat types (such as *Sphagnum* moss) may go as low as 0.04 g cm^{-3}. Total porosity (or percentage of total pore space per volume) is very high and ranges between 80 % and 95 % (Hobbs, 1986; Baird and Waldron, 2003), although effective porosity (or percentage of interconnected pore space per volume) tends to be low and ranging from 10 % to 40 %

(Ours et al., 1997). Hydraulic conductivity in peat tends to range widely from the orders of 10^{-2} to 10^{-7} cm s^{-1} (Chason and Siegel, 1986). The high organic matter content that can exceed 98 % (Hobbs, 1986) results in large specific surface area (up to 200 m^2 g^{-1}) and high cation exchange capacity (CEC) ranging from 24 to 180 meq 100 g^{-1} (Bunt, 1988) and dominated by major metal cations (Ca^{2+}, Mg^{2+}, K^+, Na^+).

The composition of peat is complex (due to the presence of organic matter in various stages of decomposition) with lignin and cellulose (major components of the cell walls of plants) as major constituents. Although some controversy still exists, lignin is thought to be the precursor of humic acids in peat. Humic substances (including humic acids and fulvic acids) typically reach 22 % content in peat soil (McDonnell et al., 2001). These components, and especially lignin, contain polar groups such as alcohols, aldehydes, carboxylic acids, phenolic hydroxides, and ethers that can be involved in chemical bonding (Bailey et al., 1999). Peat can attenuate the movement of contaminants from groundwater by adsorption (or the process by which molecules are attracted and retained on the surface of peat molecules). Due to the polar character of peat, adsorption is very effective on dissolved solids such as metals, organic contaminants, and divalent cations (Brown et al., 2000). These processes, inducing sorption in peat, are also a subject of controversy. Theories to explain how metal ions are sorbed onto peat include ion exchange, surface adsorption, chemisorption (or adsorption by chemical forces instead of physical), and complexation (or formation of complexes, structures conformed by a metal atom surrounded by a number of negatively charged ions, Brown et al., 2000). It is commonly accepted that ion exchange and complexation are the most prevalent of these mechanisms, and they are dependent on conditions such as pH, ionic strength, peat type, and concentration (Bailey et al., 1999). Solute retardation has been largely used to explain sorption processes in peat, while the properties of the peat matrix and its contribution to retardation have been commonly ignored (Hoag and Price, 1997).

Types of peat and peat accumulating systems

Definitions of peat are generally not very strict in terms of organic content requirements, and although it is generally accepted that most peats contain less than 20 % inorganic matter (i.e., unburnable), accumulations of poorly decomposed vegetable matter are still considered peat with values up to 35 % inorganic matter or even 55 % inorganic matter (in certain commercial operations). According to the USDA (United States Department of Agriculture), organic matter can be classified as (1) peat (or fibric), if virtually all remains are intact enough to allow identification of plant forms; (2) muck (or sapric), if decomposition of remains is enough to limit recognition; and (3) mucky peat (or hemic), if remains can be partially recognized. Furthermore, peat can be classified according to its origin as (1) sedimentary peat formed from floating aquatic plants such as algae; (2) moss peat formed from mosses and including *Sphagnum*; (3) herbaceous peat formed from herbaceous plants such as sedges, reeds, and cattails; and (4) woody peat formed from woody plants such as trees and shrubs.

Peat soil accumulations result in the formation of peatlands, wetland terrains characterized by a naturally accumulated layer of peat and dominated by peat-forming plants. Classification and definition of wetlands have been traditionally difficult and controversial mainly due to the lack of international consensus, with few attempts to establish a global system only lead by the Ramsar Convention on Wetlands of International Importance (Scott and Jones, 1995). According to the US Environmental Protection Agency (EPA), wetlands are defined as "areas where water covers the soil, or is present either at or near the surface of the soil all year or for varying periods of time during the year, including during the growing season," and can be classified as (1) marshes or continually water-inundated wetlands characterized by emergent soft-stemmed vegetation adapted to saturated soil conditions; (2) swamps or wetlands dominated by woody plants; (3) bogs or peat-forming wetlands (peatlands) characterized by spongy peat deposits, acidic waters, and a floor covered by a thick carpet of *Sphagnum* moss that usually receive all or most of their water from precipitation rather than from other sources (e.g., runoff, groundwater, or streams), therefore resulting in low presence of nutrients for plant growth (ombrotrophic); and (4) fens or peatlands that receive nutrients from sources other than precipitation (minerotrophic) that tend to be less acidic and present higher nutrient levels as compared to bogs.

Bogs are very low productive ecosystems when compared to other peatlands such as fens. Peat is a heterogeneous material controlled by highly complex spatial and temporal gradients. The two main factors responsible for these gradients (water budget and water quality) are strongly interrelated to the controls on productivity and vegetation type in peatlands: pH, available nutrients, mineral concentration, CEC, and water-table position. Rainwater nourishing bogs are usually very acidic and low in nutrients (e.g., N, P, K, Ca, Mg). Mineral content of bog plants is lower than in plants from fens or swamps (Archibold, 1995). Bogs are characterized by low water pH and high organic content of peat (high capacity to absorb ions). The rate of water flow controls the supply of nutrients to the plants and affects their rate of growth (Clymo, 1973). Water level variations also induce vegetation gradients, by shifting from wet- to dry-tolerant plant species. Most of the nutrients present in peat soils are not available for plant growth due to their previous incorporation into organic compounds. Soluble elements (such as Ca) are washed out and may accumulate in the underlying peat. Other elements (such as K) are lost through litter (uppermost layer of organic debris) formation (Archibold, 1995). Nitrogen enters bogs mainly through wet and dry atmospheric deposition and is particularly low in wetland ecosystems (Morris, 1991). Vegetation in bogs has adapted to these nutrient-deficient habitats. For example, vascular plants and bog shrubs in bogs withdraw nitrogen and other species (e.g., P, K) out of organs and foliage before it is shed (Archibold, 1995). Microbial activity is also poor in bogs due to the poor substrate induced from low nutrient levels, which reduces microorganisms in acid peat. Organisms that feed upon fungi and bacteria are therefore also restricted.

Geographic distribution

Although a global wetland inventory still remains incomplete, total wetland area is currently accepted to be between 8 and 10 million km^2 or 6–8 % of the land surface of the earth (Lehner and Döll, 2004). Wetland distribution is roughly bimodal with approximately 50 % located in boreal and arctic regions (e.g., 50 to 70ºN, dominated by boreal peatlands) and about 35 % located in tropical/subtropical regions (between 20ºN and 30ºS, dominated by swamps and marshes) (Matthews, 2000). In terms of peatlands, total area is estimated at about 3 % of the earth's land surface (or about 4 million km^2) (Gorham, 1991). Peat formation is primarily dictated by climate, and therefore peatland distribution is mainly concentrated across specific climate regions. Therefore, peatlands are particularly abundant in cold (boreal and subarctic) and wet (tropical/subtropical and oceanic) regions as described earlier. This geographic distribution can be explained by two main reasons: (1) water is the most important factor controlling peat accumulation and (2) temperature directly affects the production and decay of organic matter (Parish et al., 2008). The presence of large peatlands such as western Siberia, the Hudson Bay Lowlands, or the Amazon Basin is also a consequence of large flat surfaces that allow water logging conditions. North American peatlands represent about 45 % of global peatland area, followed by peatlands in Asia (about 37 %), Europe (12 %), South America (4 %), Africa (1.4 %), Oceania (0.1 %), and Antarctica (0 %) (Parish et al., 2008).

Uses

Peatlands (and wetlands in general) are one of the most important natural ecosystems in the world and recognized to perform many ecological functions. Wetlands are among the most productive ecosystems, comparable to rain forests and coral reefs. The US Fish and Wildlife Service estimates that 43 % of endangered species relies on wetlands for survival. Peatlands provide numerous benefits, such as unique wildlife habitats, biodiversity conservation and climate regulation, natural mechanisms for water purification, flood storage, recreational opportunities, and natural products for societal use. They are important in the biogeochemical cycling of nutrients, providing the necessary conditions for nutrient removal from surface waters. Peatlands account for about 33 % of terrestrially stored soil carbon, and they are hence directly related to the global carbon cycle and impact greenhouse gas concentrations, such as methane (CH_4) and carbon dioxide (CO_2) in the atmosphere (Gorham, 1991). For example, emissions from northern peatlands have been estimated to account for approximately 7 % of the global annual emission of methane to the atmosphere (Khalil, 2000).

Common peatland uses include (Parish et al., 2008) (1) agriculture, such as grazing and growing crops, often resulting in drainage and subsequent problems of subsidence, fire, and soil deterioration; (2) forestry such as timber harvesting; (3) peat extraction, commonly for fuel; (4) water regulation, acting as water reservoirs for drinking water; (5) biodiversity, including specialized organisms that are adapted to the unique conditions; (6) research, education, and recreation acting as archives and often important for tourism; and (7) carbon storage, containing nearly 30 % of all carbon on the land.

Environmental importance and climate change

Peatlands are important global stores of carbon and a critical component of climate regulation; however, their response to global warming is still a major uncertainty in climate modeling (Wania et al., 2004). Peatlands have absorbed about 1.2 trillion tones of carbon dioxide over the past 10,000 years inducing a net cooling effect on the earth. Recent studies, however, indicate that degradation and drainage of peatlands has converted many peatlands from a net store to a source of carbon emissions (Parish et al., 2008). Although the fate of peatlands when subject to changes in global temperature and rainfall regime is still uncertain, most predictions agree that changes will have a negative impact on peatlands by accelerating rate of degradation and release of stored carbon (IPCC, 2007). Such degradation will likely result in an increase in greenhouse gas emissions affecting millions of people around the world the way it is already occurring in Southeast Asian peat swamp forests after drainage and fires or from the threatening effects on the water and food supply after the destruction of peatlands in Africa, Asia, or Latin America.

Although present coverage represents less than 3 % of the earth's land surface, C storage in peatlands is estimated between 270 and 370 Tg C, approximately amounting for 35–50 % of the total 796 Tg C held in the atmosphere as CO_2 (IPCC, 2007). Tropical peatlands show the highest uncertainties from all estimates, with C stores ranging between 8 and 258 Tg C (Hooijer et al., 2006). Peatlands are also important sources of atmospheric methane, but methane-producing bacteria in the deeper parts of the bog are postulated to generate a reservoir of free-phase methane (Romanowicz et al., 1995). Recent work suggests that the mass of free-phase gas in peatlands is considerably greater than that in the dissolved phase (Fechner-Levy and Hemond, 1996). Correlation between CH_4 emissions and water-table elevation (Roulet et al., 1993), or the reduction of water flow due to the pore space blocking by biogenic gas bubbles in peat soils (Beckwith and Baird, 2001), is indicative of the close connection between carbon cycling and hydrological processes. Emission occurs via exchange through vascular plants, as well as episodic ebullition (bubbling out) events that can release large volumes of gas over a short time scale (Fechner-Levy and Hemond, 1996). Ebullition primarily depends on the volume of free-phase gas in the peat, as well as the amount of dissolved gas in peat pore waters available to replenish free-phase gas during ebullition losses (Rosenberry et al., 2003). Glaser et al. (2004) concluded that ebullition of gas from deep peat may represent a large, hitherto unrecognized source of radiocarbon depleted methane emissions from northern peatlands. These natural greenhouse emissions must be estimated in order to determine the relative significance of anthropogenic emissions to global climate change (Rosenberry et al., 2003), particularly since it is well accepted that peatlands have contributed to the variable concentrations of greenhouse gases in the atmosphere though the last 15,000 years (Parish et al., 2008). This is especially critical when considering that climate change effects are already visible in some peatlands though the melting of permafrost, changing of vegetation patterns in temperate peatlands, or desertification of certain peatlands.

Summary

Peat soils present very unique physical and chemical properties particularly in terms of porosity, water-holding capacity, and organic matter content. Furthermore, peat soils and peatlands are the most efficient terrestrial ecosystems storing carbon, efficiently sequestering and accumulating carbon for thousands of years. Peatlands are very important for biodiversity conservation and water regulation. Field studies related to peat hydrology and ecology are extensive during the last century (Weber, 1902, 1908; Boatman and Tomlinson, 1973; Glaser et al., 1981; Clymo, 1984; Hobbs, 1986); however, many aspects of peatland science are still very much uncertain. In that regard, although some controversy exists regarding the effect that changes in global temperature and rainfall regime (i.e., global warming) may induce in peatland's carbon storage, most predictions foresee a negative impact due to the accelerating rate of degradation in peat and the release of stored carbon in the form of greenhouse gases. Past climate changes over the last 15,000 years visible in the soil record support this conjecture. Furthermore, some current changes are already apparent as well in certain peatlands such as those affected by permafrost melting or desertification. Given the projected major changes in temperature and precipitation anticipated under climate change scenarios, it is critical that we achieve a better understanding of the effects of such changes on the carbon dynamics of peat soils and particularly in terms of greenhouse gas emissions to the atmosphere.

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Cross-references

Anaerobic Environments
Carbon Sequestration
Climate Change
Wetlands

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pH

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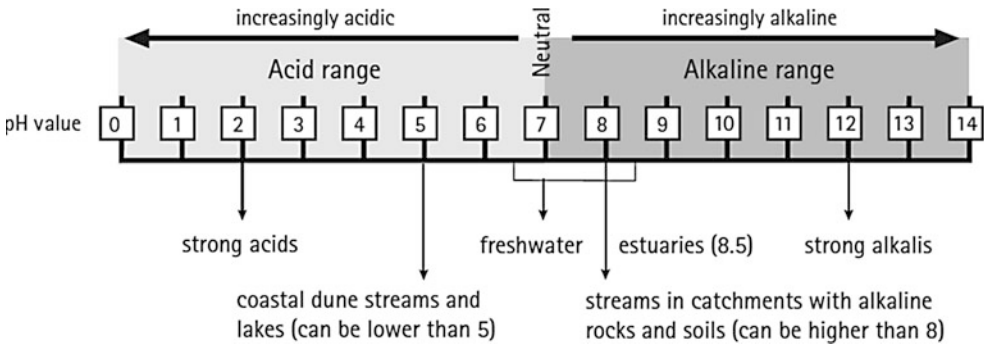
Synonyms

Hydrogen ion concentration

Definition

pH is an expression of the intensity of the basic or acid condition of a liquid. Mathematically, pH is the logarithm (base 10) of the reciprocal of the hydrogen ion concentration (Ecological Dictionary, 2011).

The pH scale ranges from 0 to 14 (Figure 1). Pure water has neutral pH very close to 7 at 25 °C, but when chemicals are mixed with water, the mixture can become either acidic or basic. A pH less than 7 is acidic, and a pH greater than 7 is basic or alkaline (EPA, 2011). The Canadian Water Quality Guideline for the Protection of Aquatic Life recommends that the pH of marine and estuarine waters should be maintained between 7.0 and 8.7 pH units (Locke, 2008). However, according to the Australian and New Zealand Environment and Conservation Council guidelines, the acceptable pH range for an estuary is between 7 and 9 (NSW, 2010). Since water pH can be affected by chemicals, pH is an important indicator of water quality that is changing. Pollution can change a water body's pH, which in turn can harm the animals and plants living there (USGS, 2013). The pH is often described as a "master variable" in aquatic systems because many properties, processes, and reactions occurring in these systems are pH dependent (George et al., 2012). The pH of water determines the



pH, Figure 1 pH guide (AGDEH, 2006).

solubility and biological availability of chemical constituents such as nutrients and heavy metals (Ecology, 2013). Estuarine pH values change in response to natural biological activities such as plant photosynthesis. During this process, plants remove carbon dioxide (CO_2) from the water and expel oxygen (O_2) which is essential for aquatic flora and fauna respiration (AGDEH, 2006). The removal of CO_2 results in a higher pH, and the water becomes more alkaline. When algae naturally begin to increase in estuaries during the spring, pH levels tend to rise. An overabundance of algae (called an algal bloom) may cause pH levels in an estuary to rise significantly, and this can be lethal to aquatic animals (NOAA, 2012).

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PHARMACEUTICALS

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Synonyms

Therapeutic drugs

Definition

Pharmaceuticals are a class of contaminants that are extensively and increasingly being used throughout the world in human and veterinary medicine. They encompass all prescription, nonprescription, and over-the-counter human therapeutic drugs used for the diagnosis, treatment, and prevention of disease, illicit or recreational drugs, products of personal care (e.g., fragrances, lotions, cosmetics, and sunscreens), veterinary drugs, and bioactive food supplements (Khetan and Collins, 2007).

Sources and legacy

Advances in sample pretreatment and analysis have allowed the identification of trace quantities of pharmaceuticals in all aquatic compartments in the last two decades. The widespread occurrence of these substances are of concern because they are designed to perform a specific biological effect; thus they have many of the necessary properties to provoke effects in nontarget species in the aquatic and continental ecosystems at lower concentrations than the intended therapeutic effect (Halling-Sorensen et al., 1998; Fent et al., 2006). It is also now emerging that pharmaceuticals and their metabolites can bioaccumulate and may have significant, but largely unstudied, consequences for individuals, populations, and ecosystems (Sumpter, 2009; Arnold et al., 2013). Monitoring the levels and fate of these contaminants in the marine environment, nevertheless, is a huge challenge due to the very low levels these compounds occur in the environment and the ever-expanding range of pharmaceuticals.

Pharmaceuticals can be categorized according to their therapeutic effect: antibiotics, anti-inflammatory drugs, analgesics, lipid regulators, beta blockers, steroids and related hormones, cancer therapeutics, antiepileptics, antidepressants, diuretics, tranquilizers, and X-ray contrast (Daughton and Ternes, 1999; Nikolaou et al., 2007). Among these compounds, caffeine is the most widespread pharmaceutical (Bradley et al., 2007), whereas diclofenac, clofibric acid, acetaminophen, ibuprofen, aspirin, carbamazepine, atorvastatin, gemfibrozil, fluoxetine, and 17β-ethynylestradiol have also become ubiquitous in the environment (Arnold et al., 2013; Liang et al., 2013).

The consumption of pharmaceuticals around the world has been rising consistently with the advances in medical technology and growing health care spending (Khetan and Collins, 2007). After usage, pharmaceuticals are excreted in their original form or as metabolites and may enter aquatic systems via different ways. The main pathway for pharmaceuticals is, therefore, domestic sewage treatment plants, which are not designed to remove trace compounds of pharmaceuticals from influents. Some techniques like photocatalytic degradation (Doll and Frimmel, 2004), advanced oxidation, and reduction processes (Jeong et al., 2010) or activated powdered carbon (Lipp et al., 2012; Margot et al., 2013) seem to be suitable tools for removal of some groups of pharmaceuticals.

Nevertheless, they are not always affordable, especially for developing countries. Additionally, there are many places around the world where sewage treatment plants are still not available and domestic sewage is discarded untreated directly into the environment, including in estuaries. Leachate from landfills, inadequate disposal of unused medicines, and the wash off of personal care products are also important sources of pharmaceuticals.

The use of pharmaceuticals in hospitals and households are quite different, once some drugs (e.g., certain antibiotics) are only used in hospitals, although others only prescribed by practitioners (Kümmerer, 2001). The predicted concentrations of antibiotics in hospital effluents are in the range of the semi-maximum inhibitory concentration (MIC_{50}) of sensitive pathogenic bacteria for some active substances; hence the development of antibacterial resistance in the environment cannot be excluded (Kümmerer, 2001).

Many animal operations, including livestock and aquaculture, generate manure that also contains antibiotics and other drugs (Liu et al., 2012; Liu et al., 2015; Steele et al., 2013). Veterinary drugs, comparing to human pharmaceuticals are more likely to directly contaminate soil and groundwater. The most important routes of entry of these pharmaceuticals into estuarine environments are likely to be the direct discharge of aquaculture products, the excretion of substances in urine and feces of livestock animals, and the wash off of topical treatments (Boxall et al., 2003).

A striking difference between pharmaceuticals and other contaminants with respect to environmental release is that pharmaceuticals have the potential for ubiquitous direct release into the environment worldwide, anywhere that humans live or visit (Daughton and Ternes, 1999). Nevertheless, the behavior and fate of pharmaceuticals and their metabolites in the coastal systems is not known. The low volatility of pharmaceuticals suggests that their distribution will occur through aqueous transport and interactions with suspended particulate materials, as well as via food web transfer. Several classes of pharmaceuticals seem to have wide range of persistence in aquatic environments, and some are highly persistent (Cuong et al., 2011).

Environmental screenings were conducted on groundwater, surface water, treated wastewater, and even coastal waters (e.g., Halling-Sorensen et al., 1998; Kümmerer, 2001; Fent et al., 2006; Lisboa et al., 2013; Stamatis and Konstantinou, 2013; Valley et al., 2014). Evidences have already been found demonstrating relationships between the presence of certain pharmaceuticals and their toxicity in aquatic biota and humans (Daughton and Ternes, 1999). Acutely toxic levels of pharmaceutical compounds were identified in laboratory tests using algae, zooplankton, and fish (Fent et al., 2006). Existing knowledge about residues of pharmaceuticals in aquatic systems indicate that they are unlikely to pose a risk for acute toxicity (Cuong et al., 2011). Recent studies are focusing on evaluating the biological effects of mixtures of pharmaceuticals that are mechanically linked or chemicals from different classes that may cause drug-drug interactions (Arnold et al., 2013; Metcalfe, 2013). However, the environmental significance of pharmaceuticals in estuaries remains poorly understood and there is a paucity of information on their fate in the environment. Further investigations about the long-term fate and impacts of pharmaceuticals in coastal ecosystem are needed.

Summary

A wide range of pharmaceuticals including analgesics, antibiotics, and stimulants are detected in the environment, including coastal ecosystems. The major sources of these contaminations are municipal and hospital wastewater treatment plants. Today, pharmaceuticals can be found, at trace levels, widely in a variety of environments, including estuaries. Many of them show persistence to biodegradation in the environment; therefore they may bioaccumulate in living organisms potentially causing adverse effects.

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Cross-references

Ecological Monitoring
Nonpoint Source Pollution

PHI SCALE

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Synonyms

Phi grade scale

Definition

The phi scale is a sediment particle size scale, defined as a logarithmic transformation of the geometric Udden-Wentworth grain size scale (AGI, 2013). The phi diameter is calculated as the negative logarithm to the base 2 of the particle diameter (in millimeters).

Discussion

The phi scale was introduced by Krumbein (1934, 1938) as a convenient means of visualizing and statistically analyzing sediment grain size distributions over a wide range of particle sizes. The earlier Udden-Wentworth scale is a millimeter-based scale, with an unchanging ratio between class limits. The phi scale allows more emphasis for the finer grain sizes. Phi size values for the sediment class limits range from −5 phi (for a diameter of 32 mm, or very coarse pebble size) down to +10 phi (for a diameter of 1/1,024 mm, or clay size). The scale was developed specifically as a statistical device to permit the direct application of conventional statistical practices to sedimentary data.

The phi scale has distinct advantages over geometric size scales. Tanner (1969) summarized some of the advantages as follows: geometric basis; equal spacing of size class divisions, with integers representing the limits of the Wentworth size classes; simple nomenclature; potential for fine subdivisions; inclusion of a wide range of grain sizes; wide acceptance; a close match of boundaries between the three important “super classes” (gravel, sand, silt + clay) with the widely recognized deficiencies at about −1 phi (2 mm) and +4 phi (1/16 mm); ease of plotting on ordinary probability axes; adaptability of phi units to a variety of sophisticated statistical procedures; ease of interpretation of non-Gaussian data; and ready availability of sieve screens in standard phi sizes.

Due to its nonlinear and dimensionless nature, the phi scale presents some challenges in reporting, especially outside the field of sedimentology. As a consequence, results are often reported in both phi units and metric units. Care must be taken in such cases to stipulate how the data were transformed.

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Cross-references

Estuarine Sediment Composition
Sediment Grain Size
Sediment Sorting

PHOSPHORUS

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Definition

Phosphorus (P) is an essential chemical element for aquatic life. It has an atomic number of 15, and its atomic mass is 31 u.

Sources

Phosphorus enters rivers, streams, and other watershed influent systems through the weathering of phosphoric rock, and it is then transported to estuaries. Industrial, agricultural, and domestic effluents are additional sources of this element (Bianchi, 2007). Detergents commonly include phosphates as a component; domestic sewage therefore generally carries high levels of phosphorus to coastal aquatic systems. Phosphorus can reach elevated concentrations in estuaries, where it can compromise water quality. In estuarine ecosystems, phosphorus is rarely encountered in molecular form but occurs as phosphate consisting of a phosphorus atom bonded to four oxygen atoms ($PO_4{}^{3-}$). The phosphorus cycle in estuaries is regulated in large part by the activities of microbes (Day et al., 2012).

Essential Nutrient

Phosphorus is essential for the growth of autotrophs, which transform orthophosphate into mononucleotide acids (ATP, ADP), phospholipids, fatty acids, and proteins (structural or genetic information for proteins such as DNA and RNA) during photosynthesis (Day et al., 2012). It enters the trophic web through primary consumers. Organic matter regeneration, ion exchange, and re-suspension processes maintain high levels of bioavailable dissolved phosphate (Bianchi, 2007). The optimal nitrogen:phosphorus ratio (N:P ratio) for phytoplankton growth is 16:1 and is called the Redfield ratio (Redfield et al., 1963). N:P ratios in estuaries are commonly lower and/or higher than the predicted Redfield ratio (Bianchi, 2007).

Eutrophication

In estuaries, an excess of phosphorus can be problematic, often resulting in eutrophication under appropriate light and temperature conditions, which can impact the structure and function of these ecosystems. For this reason, phosphorus is employed as a chemical indicator of the health of estuaries, and it is usually monitored in water quality surveys.

Forms

Phosphorus is often available as dissolved inorganic phosphorus (DIP), dissolved organic phosphorus (DOP), and particulate organic phosphorus (POP). DIP is rapidly assimilated by algae during the photosynthetic process. In aquatic ecosystems, the orthophosphate ion ($PO_4{}^{3-}$) is the most oxidized and stable compound. The phosphorus cycle is relatively simple compared with that of the nitrogen cycle (Day et al., 2012), because phosphorus has no gas phase and is less subjected to biological transformation. In estuarine environments, DIP is highly reactive and forms complex compounds with metals, including iron and manganese (oxides and hydroxides), which form insoluble precipitates under aerobic conditions. However, it may return to the dissolved form when the redox conditions are sufficiently reducing. Due to the high charge on this ion, it may be fixed on particle surfaces, including clays and organic detritus that act as cations. Such sorption-release processes comprise a buffering mechanism for DIP in estuaries, maintaining DIP bioavailability such that it is not considered a limiting nutrient (Bianchi, 2007).

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Cross-references

Algal Blooms
Anthropogenic Impacts
Ecological Stoichiometry
Eutrophication
Macronutrients
Nonpoint Source Pollution
Nutrient Dynamics

PHYSIOLOGICAL MECHANISMS

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Definition

Physiological mechanisms refer to the physiological adaptations needed for life in estuaries and coastal lagoons.

Introduction

Perhaps the most important specialization needed by an estuarine organism is the ability to live in brackish water of varying salinities. Living sometimes in rapidly changing external media, animals are also exposed to wave activity, high silt levels, freshwater inputs, and temperature extremes. The ability to avoid adverse conditions or regulate their ionic and water balance allows species to occupy habitats where they are able to grow rapidly and exploit readily available food sources (Lockwood, 1976). Being close to human settlements, estuaries and their brackish-water organisms are subject to anthropogenic inputs from industry, fishing, and waste discharges, including untreated and treated sewage. These can result in highly elevated nutrient levels, eutrophication, and hypoxic sediments. Many estuaries are contaminated with trace metals, PCBs, and other substances, and the combinations of stressors can affect all aspects of the biology of estuarine organisms. Adaptations include morphological, behavioral, and physiological mechanisms that allow animals to evade the stresses, those which regulate osmotic and ionic balance, and specialist mechanisms to survive in hypoxic conditions. Estuaries are under pressure from climate change, and one can ask whether the organisms that live in this habitat have physiological mechanisms that allow them to adapt to changes in water levels and ocean acidification.

Adaptation to salinity changes: osmoconformers and osmoregulators

Brackish-water organisms are euryhaline, tolerating a wide salinity range (0.5 to 35 psu). Some are conformers, species where the body fluids and cells are usually equal in osmotic pressure (isosmotic) to the external environment, and others are osmoregulators where the body fluids are regulated homeostatically, maintaining a constant body fluid concentration despite changes in the external environment. As shown in Figure 1 (from Willmer et al., 2005), there are varying levels of osmoconforming and osmoregulating among different animals. Normal seawater is 34 psu, yielding about 1,000 mOsm, and the figure shows the variation of internal body fluid with external salinity for a range of brackish-water species. Although there is no single pattern of osmoregulatory ability of estuarine organisms, three main mechanisms are involved in the adaptation to osmotic changes. These are (1) altering external permeability, (2) varying salt uptake, and (3) conducting cellular osmoregulation (see Willmer et al., 2005).

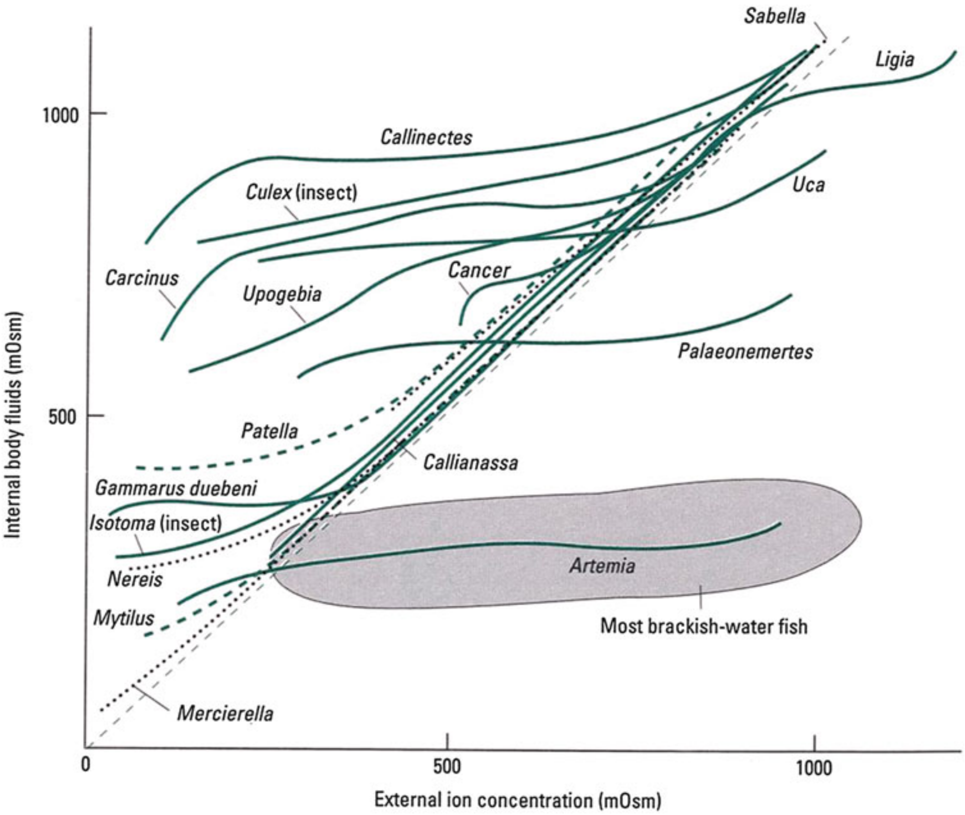
Polychaetes and mussels, which are generally thought of as osmoconformers, have limited or no osmoregulatory control; they are successful in estuaries because they tolerate a wide range of internal concentrations and tissue volume changes, although they may possess some intracellular isosmotic regulation using organic solutes compatible and counteracting organic osmolytes. Osmoconforming is a good mechanism because it avoids energy use in active transport, and this lowers the metabolic costs and reduces food requirements.

Fish and marine vertebrates are osmoregulators and are able to maintain a steady internal osmotic pressure regardless of osmotic changes in the external environment. In fish, osmoregulatory organs are the digestive tract, gill, kidney, urinary bladder, and liver in ureotelic regulators. Some crustaceans, such as the brine shrimp *Artemia*, are also osmoregulators, living in pools of varying salinities and maintaining a low internal osmotic pressure. They drink large amounts of water (8 % of body weight per day), retain water with a highly impermeable exoskeleton, and excrete excess ions with special salt glands on their gills.

Bony fish (teleosts) and higher vertebrates are hypoosmotic, maintaining a low cellular osmolarity between 250 and 400 mOsm regardless of the external salinity. This adaptation allows them to migrate between the sea- and freshwater for spawning and feeding. They tend to lose water and gain excess salts through their external surfaces. One mechanism to cope with this problem is for marine fish to drink seawater. The digestive tract restricts some ions, but some are imported allowing water to follow. Excess salts are carried in the blood to the gills where specialized epithelial chloride cells actively transport NaCl outward. Salts are lost in the feces, and there is very little urine production, again reducing the water loss. A more detailed description of the mechanisms used in other marine vertebrates is given by Sherwood et al. (2005).

Cartilaginous (elasmobranch) fish, sharks, and rays are different. They maintain high osmotic pressures often above their external medium by producing urea which is stored throughout the body. Urea is toxic and such concentrations would be fatal to many organisms. These fish, however, are peculiar in that they have evolved specialized structures that are urea resistant and they also produce methylamine osmolyte (TMAO), with concentrations about half of that of the urea, that counteracts the destabilizing effects of urea on protein structure and function.

Crustaceans have a hard relatively impermeable external surface, and there are interspecific differences in osmoregulatory ability between species. Some are osmotic and ionic conformers in full strength seawater



Physiological Mechanisms, Figure 1 Variation of internal body concentrations with external salinity for a range of brackish-water species. Arthropods are shown by *solid lines*, mollusks by *dashed lines*, and worms by *black dotted lines* with teleost fish represented by the tinted area (From Willmer et al., 2005).

and regulators usually below 26 psu (Henry, 2005). Crustaceans, such as the blue crab *Callinectes sapidus*, could be described as strong osmoregulators, able to maintain high gradient of 600 mOsm internal osmotic pressure when acclimated to freshwater in the laboratory. While in the past, species have been classified as either weak or strong hyperosmoregulators. Henry et al. (2012) suggest that they should be separated into three groups based on mechanisms of transport and based on gill epithelia which have either low or high electrical conductance. Euryhaline osmoconformers typically would be those with very high conductance, no active transport of NaCl, and hyperosmoregulators which have active NaCl uptake and either high conductance (leaky) gills or low conductance (tight) gills. Henry et al. (2012) provide a full description of the mechanisms.

Tolerance to environmental extremes

Salinity, temperature, and water loss

Salinity and temperature are among the most important factors affecting the metabolic rate, heart rate, and growth of estuarine organisms (Newell, 1979; Marsden, 2004). Many estuarine invertebrates survive in a wide range of salinities, and while responses are species specific, they depend on the exposure time, salinity level, and also the exposure temperature. Some of the most tolerant species include pulmonate snails, *Amphibola crenata* and *Melampus bidentatus*, and high-level salt-marsh species, which are highly tolerant of desiccation and temperature extremes (Shumway and Marsden, 1981). Some crustaceans and mollusks acclimate to new salinities, and survival of many estuarine invertebrates depends on both temperature and salinity (e.g., the eastern oyster *Crassostrea virginica* (Heilmayer et al., 2008), the sand shrimp *Crangon crangon*, and the shore crab *Carcinus maenas* (Jillette et al., 2011)). In well-oxygenated waters, mortality is minimal at temperatures from < 5 °C to 22 °C and salinities from 15 to 30 psu. When shrimp were exposed to poorly oxygenated water, mortality was much higher. Shrimp survived best at lower temperatures and salinities slightly higher than normal seawater (Haefner, 1970). The salinity tolerance will change during development, with salinity tolerance increasing from egg, larval stages, through later life history stages (Charmantier et al., 2001).

Heat stress

Temperature plays a critical role in regulating the metabolic rate of organisms as well as affecting population parameters, such as population size and distributions (Portner et al., 2008). Numerous studies have investigated the thermal lethal tolerances of estuarine ectotherms, and results suggest that species inhabiting the upper shore have a higher thermal tolerance, but also live closer to their thermal maxima, than those living at lower tidal levels. In more recent studies, Madeira et al. (2012a) used behavioral endpoints to determine the critical thermal maximum (CTMax) and found results which were consistent with some previous studies. They confirmed lower CTMax for subtidal, demersal, and northern species exposed to colder temperatures. While some previous studies have suggested that the same species from different localities might differ in thermal tolerances, this was not found for those investigated in this study. Also, while it was previously thought that there would be taxonomic differences in thermal tolerances, this was not apparent in the study, and taxa from the same habitats had similar thermal tolerances, suggesting species have evolved similar stress responses. It is predicted that the species most vulnerable to increased seawater temperatures are those from thermally unstable environments. This includes some estuarine habitats, although species that live here are expected to have a high CTMax, have limited powers of acclimation, and live close to their thermal maximum.

Respiratory adaptations

Dissolved oxygen is readily available in open coastal marine environments, but in estuarine waters it is often low, being described as hypoxic. Estuarine waters may also exhibit hypercapnia or increased levels of carbon dioxide. In aquatic organisms, the gills are multifunctional organs, important not only in oxygen uptake but also in ion transport which underlies hemolymph osmoregulation, acid–base balance, and ammonia excretion. They are also the site of uptake of trace metals. When exposed to air, the gills collapse and adhere together making gas exchange difficult. Marine muds may be nutrient rich, and the sediment becomes blackened close to the surface where bacteria have converted nitrates and sulfates to nitrites and sulfides. In this habitat estuarine organisms use aerobic and anaerobic respiration and have structural and behavioral adaptations to minimize exposure to environmental stressors.

Because diffusion alone cannot deliver sufficient oxygen and carbon dioxide between external respiratory surfaces, tissues, and mitochondria, animals have evolved respiratory structures and circulatory systems to transport respiratory gases around the body. Many taxa also use respiratory pigments which increase their ability to pick up and deliver oxygen to the tissues. These are similar in structure, but hemoglobin and chlorocruorin, which are found in many estuarine polychaetes, have a prosthetic group called a heme, a porphyrin attached to one atom of ferrous iron. Hemocyanin, found in mollusks and crustaceans, does not have porphyrin; instead the prosthetic group is a polypeptide attached to copper and sulfur rather than iron. Hemerythrin is a less common respiratory pigment, being found in three phyla (sipunculids, priapulids, and brachiopods), and the pigment contains iron but without a heme group. The oxygen-carrying capacity of the different pigments is affected by pH, and raised carbon dioxide levels shift the oxygen equilibrium curve to the right in a Bohr shift, allowing the delivery of oxygen on the venous side of the capillary networks (Wells, 1980).

Oxygen uptake in most aquatic organisms occurs across the gills, and numerous studies have investigated how internal factors such as size, sex, and activity affect the uptake. External factors also affect the rate; oxygen tension, temperature, and salinity are the most important (Newell, 1979). Temperature is a key variable and affects all of the metabolic processes, with the temperature coefficient Q_{10} representing the sensitivity to temperature and the energy demands for growth. In estuaries where temperature fluctuations may occur in a predictable way, organisms may have a temperature insensitive region of their rate–temperature curve where there is no increase in oxygen uptake with increasing temperature. This adaptation minimizes energy expenditure in stressful environments where food resources may be limited. Shrimp are usually found in high-salinity waters within estuaries, and several authors have found that temperature rather than salinity has the greater effect on metabolic rate. Spanopoulos-Hernandez et al. (2005) found that the Q_{10} of juvenile shrimp did not vary with salinity but increased markedly with temperature, with a Q_{10} of 1.88 between 20 °C and 30 °C.

In response to declining oxygen levels in water, estuarine organisms are sometimes described as either oxygen conformers, which generally decrease their metabolic rate, or regulators which are able to maintain a steady oxygen uptake rate in declining oxygen tensions. While many organisms do not fit clearly into particular categories, many species are able to maintain some degree of regulation using a combination of mechanisms including altering the ventilation or stroke rate and using temporary oxygen stores, blood pigments, and aerobic metabolism.

Ability to survive aerial exposure and desiccation

The ability to survive aerial exposure is a requirement for estuarine organisms living in the intertidal zone. Some invertebrates such as polychaetes become inactive on the outgoing tide, while barnacles and bivalves can open and close irregularly to expose respiratory surfaces to the air, which has higher oxygen content than seawater. Crustaceans and mollusks from upper tidal levels are able to maintain oxygen uptake in air, while those from lower tidal levels tend to have reduced rates of respiration when exposed to air (Newell, 1979). Because they are regularly exposed to air, many estuarine organisms have a high tolerance of water loss, being able to survive up to 70 % total body loss. Shell closure and reduced metabolic rates also minimize energetic costs. Avoidance also is a good strategy for mobile invertebrates, snails and crabs, which form burrows, providing them with water, shelter, and protection from predators. Polychaetes and bivalves are invertebrates that are able to respire anaerobically, and during this period, the metabolic rate is usually greatly depressed.

Hypoxia and anaerobic metabolism

Survival in low-oxygen conditions is an important adaptation for estuarine organisms, where the tolerance is affected by numerous factors, including temperature, light intensity, body weight, pH, and feeding conditions. For example, in the shrimp *Litopenaeus vannamei,* which lives at 1–2 to 40 psu, tolerance to hypoxia is greatest when it has been cultured at the optimal salinity for growth (Zhang et al., 2006).

Many animals can simultaneously respire both aerobically and anaerobically, but the ability to respire anaerobically is critical for many estuarine polychaete worms and bivalves. As in the case of aerobic respiration, anaerobic respiration depends on having a suitable substrate to provide energy, but it has a lower yield and produces end products which need to be stored or excreted later. While some animals, fish, and crustaceans use glycogen for energy production, this results in the accumulation of large amounts of lactate. Many brackish-water organisms use an alternative anaerobic pathway, incorporating amino acids, such as arginine, which yields similar amounts of energy but has a different end product, and octopine, which may interfere less with particular enzyme systems. Bivalve mollusks and gastropods are particularly resistant to anoxia because they are able to establish homeostasis once oxygen becomes available, and they can avoid the damaging effects of rapidly increasing oxygen radicals, which lead to oxidative stress (Freire et al., 2011a). The antioxidant defenses of mollusks include the production of enzymes (e.g., superoxidase dismutase and glutathione). During anoxia, increased activities of catalase and glutathione S-transferase (GST) have been recorded in the gills of estuarine crab *Chasmagnathus (Neohelice) granulata*. It is suggested that species from more challenging habitats, for example, those stressed by temperature, salinity, and low oxygen, show higher activities of stress indicators such as heat-shock proteins, blood cortisol levels, and antioxidant levels (Freire et al., 2011a; Freire et al., 2011b). Both increased and decreased salinities affected expression/activities of enzymes of the antioxidant system. In a comparison of two closely related crabs which have different distributions, the species from the inner estuary was expected to have high levels of antioxidant enzymes compared with crabs from the more stable marine environment. This was found to be true; the more euryhaline species had greater activity of antioxidative enzymes, the less euryhaline species exhibited activation of these enzymes when exposed to air or hypersalinity.

Growth and energetics

For many organisms, the growth patterns are controlled by both internal and environmental factors including salinity which can affect reproduction and the success of larval stages (Pechenic and Kerr, 2000; Pechenic et al., 2003). Growth rate is highly dependent on the energy budget, and changes in the growth rate as a result of salinity flux are potentially affected by the standard or basal metabolic rate, quantity and quality of the food supply, food conversion, and perhaps hormonal stimulation. For estuarine clams, growth is best at intermediate salinities and often reduced at low salinities (Carmichael et al., 2004; Marsden, 2004). For flatfish (flounders), there is an optimal salinity for early development and larval growth, but this is species specific (Smith et al., 1999; Specker et al., 1999). There is considerable evidence showing that salinity influences growth in fish (Boeuf and Payan, 2001), and optimal growth occurs at intermediate salinities (8–20 psu). Sometimes this is the result of lowered standard metabolic rate, but it may not always be the case, because growth rate may also be affected by food intake, swimming performance, and activity patterns. In salmonids, there is hormonal control of salmon smoltification involving both osmoregulation and growth. It was thought that between 20 % and 50 % of the energy budget in fish was allocated toward osmoregulation, but other studies suggest it may be considerably lower (~10 % in isolated gills of cutthroat trout) (Morgan and Iwama, 1999). This remains a subject of ongoing debate.

Behavioral mechanisms and biological rhythms

Behavioral mechanisms allow mobile estuarine organisms to evade the stresses associated with shallow water environments; shrimp and crabs typically move away from dangerous hypoxic conditions. Burrowing in soft sediments is also a key adaptation, solving all the problems associated with osmotic, respiratory, and thermal stress at low tide. Many fish have seasonal patterns of development and migrate in and out of estuaries. Some crabs enter estuaries at high tide to exploit mudflat habitats, while avoiding avian predators and stresses associated with aerial exposure. The reproductive strategies of estuarine invertebrates including polychaetes are often more complex than open-coastal species, and larval release of benthic decapod crustaceans occurs according to lunar, light–dark, or tidal cycles. Spawning is usually synchronized with particular tidal conditions and larval behavior (including vertical migrations) to ensure that juveniles move back into estuaries and settle in a habitat which provides good conditions for growth (Naylor, 1976).

In the past, considerable research has been conducted to assess the mechanisms controlling rhythmic behavior of invertebrates. Crabs have been studied in detail, and research suggests that the behavior is controlled by a combination of external and internal (endogenous) factors, involving an internal clock (Naylor, 2001). Estuarine crabs held in constant conditions in the laboratory in the dark continue to display locomotor rhythms correlated with the tides and the moon with a periodicity of 12.4 h which occurs twice in each lunar day (24.8 h).

These rhythms, which are cued by tidal variables including hydrostatic pressure, salinity, and temperature, most likely have a molecular basis and have obvious adaptive value in reproduction.

Circatidal rhythms are well known in blue crabs (*Callinectes sapidus*) which undergo spawning migrations to the sea, then move closer to the mouth of the coast to release larvae. Recent research by Darnell et al. (2010) has revealed ontogenetic changes where crabs prior to oviposition had a variety of endogenous swimming rhythms (circadian, circatidal, or circalunadian), whereas ovigerous crabs from estuaries with diurnal tides had pronounced circatidal or circalunadian rhythms with swimming at the time of the ambient low tide. There were, however, differences in behavior of crabs from nontidal situations, and the authors considered whether this could be explained by genetic differences between populations. Because of high levels of genetic diversity, this was rejected, and instead the authors suggested the differences were due to the phenotypic plasticity of the clock mechanisms which can be entrained to either semidiurnal tidal, diurnal tidal, or 24 h light–dark cycles.

Effects of contaminants

Contaminants affect estuaries, partly because they are often close to cities where there are often accidental spills of sewage, oil, and industrial waste containing trace metals, chlorinated hydrocarbons, and microbial contaminants. The organisms that live in estuaries take up contaminants from the surrounding water, pore water, sediment, and their food (Marsden and Rainbow, 2004). There has been considerable research into the effects of trace metals on estuarine organisms, and, in general, crustaceans and bivalves take up trace metals, such as copper, in proportion to the concentrations in the external media (Rainbow, 1995; Luoma and Rainbow, 2008). There are few consistent patterns among different crustaceans in their responses to altered salinities, and these responses appear to be species specific and related to body size and differences in physiology. Salinity affects the speciation and bioavailability of metals in the environment, influencing their uptake by aquatic organisms (Luoma and Rainbow, 2008). Generally a decrease in salinity results in an increase in the metal content of the tissues because of an increase in free metal ion concentration, which is a result of decreased metal complexation at lower salinities. One area of current interest lies in the effects that salinity has on the physiological mechanisms in decapod crustaceans. Because copper is a functional part of the respiratory pigment hemocyanin, we would expect to find it in the hemolymph where it would be removed to other organs to be excreted or detoxified (Rainbow, 2007). In blue crabs, the anterior gills are for oxygen uptake, and the posterior gills are for ion transport. Therefore, copper, like other metals such as silver, cadmium, zinc, and mercury, can cause respiratory problems and osmoregulatory disturbances (Bianchini et al., 2008). Martins et al. (2011) recently investigated the effects of salinity on the acute toxicity of copper in the blue crab. Although there were differences in the apparent toxicity with salinity, with an increase in survival at 2 psu as opposed to 30 psu, these differences were explained simply by changes in water chemistry and not due to different physiological mechanisms at different salinities. This research showed that the gills are the key target organ for copper accumulation, and they also act as a barrier to high uptake by the hemolymph and further distribution of copper to the internal organs, including the hepatopancreas. Copper is also known to disrupt acid–base balance by restricting gas exchange in *C. maenas* and in *N. granulata*, where it results in an inhibition of carbonic anhydrase (Skaggs and Henry, 2002).

Use of biomarkers

There are multiple stressors in estuaries. Chemical contaminants, physical and anthropogenic stressors, and the effects of individual stressors can be additive, antagonistic, or synergistic (Crain et al., 2008). The search for a way to assess the overall health of animals when exposed to multiple stressors has led to the use of biomarkers and bioindicators of environmental health. Biomarkers are defined as the biological responses to environmental stressors, which include the influences of the abiotic environment and anthropogenic pollution. They can be ecological, behavioral, biochemical, physiological, chemical, genotoxic, or molecular (Scott and Sloman, 2004; McDonagh et al., 2005; Farcy et al., 2013). Some biomarkers are known as specific biomarkers; examples include metallothioneins which represent trace metal concentrations in body tissues. They are salinity dependent. Others include cholinesterase activity which is a specific biomarker for organophosphorus and carbamate pesticides and neurotoxins (Monserrat et al., 2007). Alternatively, nonspecific biomarkers (DNA damage, protein oxidation, antioxidant responses) reflect combinations of stressors (Gericitano et al., 2004). Biomarkers of DNA damage are valuable tools to assess the effects of acute and chronic exposure of aquatic animals to genotoxic substances (Amado et al., 2006). Any changes are important because they can be passed on to future generations and have ecological consequences. Rodrigues et al. (2012) assessed the effects of salinity stress on a suite of biomarkers (oxidative, energy metabolism, and neurotransmission) of C*arcinus maenas* from an estuary with very little pollution compared with one where the crabs were impacted by high chemical stress. Although crabs from the clean site showed alteration in neurotransmission and antioxidative defenses, those from the impacted site exhibited responses that were induced by both high and low salinity exposure.

Summary

Estuaries are short-lived geological features which have been subjected to changing conditions as a result of natural and anthropogenic factors. Compared with coastal ocean environments, relatively few species occur in estuaries, and they have specialized morphological, physiological, and behavioral adaptations. With global climate change, additional pressure occurs from increased sea level, elevated temperatures, and ocean acidification. Because of their shallowness, estuaries are especially susceptible to climate change, and anoxic zones that can develop are among the most threatened environments (Diaz and Rosenberg, 2008). Some estuarine organisms are able to adapt to changing temperatures by acclimation and the production of heat-shock proteins which provide cellular defense against oxidative stress (Madeira et al., 2012b). Others adapt to changing salinity regimes, and the physiological and behavioral adaptations described above would allow species to survive in hypoxic and hypercapnic environments. Thus, physiological mechanisms which have evolved over time, especially those which allow species to manage their acid–base balance when faced with changing conditions, enable estuarine organisms to survive. Fish and polychaete worms have effective mechanisms to cope with global change, but further research is needed to identify which species are sensitive to changes and those which are able to invade and exploit the estuarine environment.

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Cross-references

Aerobic Environments
Anaerobic Environments
Bioindicators
Biomonitors
Macrofauna
Tidal Flat Salinity Gradient
Water Quality

PHYTOPLANKTON

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Definition

The plant component of plankton populations in the water columns of estuaries, oceans, and freshwater habitats.

Introduction

Phytoplankton (Greek: *phyto*- plant; *planktos*- drifting) populations include a diverse representation of both microscopic eukaryote algae and prokaryote cyanobacteria. They are major oxygen producers and also represent the initial food source for numerous food webs in freshwater, estuarine, and marine waters. These taxa have a variety of shapes and sizes that include unicellular, colonial, and filamentous species (Tomas, 1997). Many of the unicellular taxa also possess spines or other protuberances that provide increased surface to volume ratios to the cells that aid in their buoyancy. Other species are motile, containing flagella allowing them movement in the water column, and are referred to as phytoflagellates. The colonial forms may include those that are enclosed in a mucilaginous by-product which is common among many of the cyanobacteria species and which allows their cells to remain clustered together. Filamentous phytoplankton may consist of straight, curved, or coiled filaments of various lengths and sizes. The phytoplankton species appear as floating entities in the water column with their distribution and transport influenced by existing currents and upwelling conditions.

Composition

Diatoms and dinoflagellates are examples of the more dominant and diverse eukaryote algal components of estuaries and are joined in these waters by a variety of species from other algal categories (Reynolds, 2007). Although a variety of the prokaryote cyanobacteria taxa are present in freshwater lakes and rivers, they have a more limited occurrence in estuaries. In general, freshwater species entering an estuary are not very tolerant to the existing salinity levels, which subsequently results in a phytoplankton community dominated by estuarine and marine species. In addition to salinity, there are other environmental variables that will influence the composition, abundance, and distribution of the phytoplankton in these waters (e.g., these include existing temperatures, nutrient availability, light intensity, predation, tidal exchange, among others). The changing environmental conditions within an estuary will generally favor a greater seasonal development and presence of several algal components, resulting in changing patterns of dominance through the year and the presence of background species. These changes result in seasonal and annual patterns of species representation and dominance. A common seasonal response associated with increasing temperatures from winter into spring is the major development of diatoms occurring in this late winter-early spring period. This growth is then typically followed by a decrease in diatom development, followed by increased concentrations among other taxa, with this sequence repeated the following year.

Size

Sieburth et al. (1978) proposed a classification system for various taxa within the plankton community based on size. These categories and the range of their sizes pertaining to the plankton constituents were picoplankton (0.2–2.0 µm), nanoplankton (2–20 µm), microplankton (20–200 µm), mesoplankton (200 µm to 2 mm), and macroplankton (>2 mm). Many of the smaller cyanobacteria are included in the picoplankton category, whereas several representative dinoflagellates and diatoms are in the meso- and macroplankton categories. For instance, among those included in the largest of the phytoplankton taxa are the dinoflagellate *Noctiluca scintillans* with a cell size of ca. 1 mm in diameter and the diatom *Ethmodiscus gazellae* having cell diameters up to 1–2 mm. However, these examples are the extreme with the majority of the phytoplankton taxa in either the nano- or microplankton size categories.

Distribution

Distribution patterns among the estuarine phytoplankton species will vary from being globally cosmopolitan to those more common to specific polar, temperate, of tropical geographic regions. Species may be introduced to other estuaries from currents within adjacent coastal waters through tidal entry, or from ballast water released from ships entering local harbors. These species may survive and eventually persist in these waters, or may perish due to conditions not conducive to their survival. It is not unusual for an invasive species to enter an estuary, survive, and become a newly established resident of that estuary.

Trophic representation

Phytoplankton may also be classified according to their trophic status. Autotrophs are the photosynthetic species, capable of synthesizing inorganic material (CO_2 and H_2O) using light or chemical energy to produce organic carbon. This is the dominant type of nutrition associated with phytoplankton represented by species possessing photosynthetic pigments (e.g., diatoms, cyanobacteria, chlorophytes). Through this process they are a major source of biomass and oxygen. The heterotrophs lack these pigments and do not conduct photosynthesis, so they obtain carbon products externally, such as by ingesting (phagotrophy) other organisms, or their organic products. This life style is common among many of the dinoflagellates which are capable of ingesting bacteria or other organisms in the water column. The mixotrophs are represented by autotrophic taxa that are also capable of phagotrophy.

Reproduction

Reproduction among the phytoplankton is primarily by binary fission, with sexual reproduction also common among several of the categories (e.g., the dinoflagellates). Included in the life cycle of many phytoplankton species is the formation of cysts, a temporary resting stage which may settle to the substrate of the estuary. Under favorable conditions these resting cysts may later develop to active developing cells that will continue growing and complete the life cycle stages of the species. Favorable environmental conditions would enhance increased cell division of these taxa resulting in greater abundance among the phytoplankton. Excessive patterns of growth will also produce high concentrations of the cells that may produce an algal bloom. Due to pigments within the cells of these algae, these blooms may impart a noticeable color to the water's surface (e.g., red tide). Based on the existing environmental conditions (e.g., available nutrients, favorable temperatures, residency time within the estuary, etc.), these blooms may be brief and involve a small area of development or be long lasting (e.g., days, weeks) and extend over a large surface area.

Environmental concerns

In contrast to phytoplankton's beneficial contributions to the waters as a food and oxygen source, they have also been associated with unfavorable conditions associated with algal bloom events. Following algal bloom development and the subsequent death of these cells, the cells undergo bacterial decomposition resulting in reduced oxygen levels in the water column, often resulting in hypoxic or anoxic conditions and accompanied by fish mortalities. This condition occurs more frequently in the smaller and shallow regions of estuaries and tidal river sections. Other bloom-forming algae also produce toxins harmful to animals and humans.

Summary

Phytoplankton represent a diverse microscopic algal community of major ecological importance in estuaries and other water-based ecosystems. They produce oxygen through the process of photosynthesis and represent a basic food source for microscopic fauna essential to various pelagic food webs. They are also recognized as frequently producing large surface concentrations of cells referred to as algal blooms that are often associated with degraded water quality conditions. Included among the phytoplankton are species that are potential toxin producers.

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Cross-references

Algal Blooms
Cyanobacteria
Invasive Species
Phytoplankton
Phytoplankton Blooms
Toxic Blooms

PHYTOPLANKTON BLOOMS

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Definition

A "phytoplankton bloom" is a major peak in biomass of planktonic algae or cyanobacteria.

Description

Despite the relatively simple character of the term "phytoplankton bloom," its applications and the meaning of the elements of the definition are in practice subject to a range of interpretations (Smayda, 1997). In most cases the term "phytoplankton" refers to photoautotrophic species that spend a significant portion of their life suspended in the water column (Reynolds et al., 2006); however, there are exceptions to this most common usage. For example, blooms can involve strictly heterotrophic dinoflagellates, such as *Pfiesteria piscicada*, or mixotrophic algae that alternate modes of growth between photoautotrophy and other nutritional modes (e.g., phagotrophy, osmotrophy) (Burkholder et al., 2008). The definition of "major" is also subject to varied interpretations. It can refer to annually recurring peaks in biomass, e.g., "spring bloom," or to events that deviate significantly from a prescribed average condition. Therefore, what is defined as a bloom in one ecosystem may not be considered a bloom in another. Even the term "phytoplankton" can be subject to variations in interpretation, such as the appearance of high concentrations of meroplankton (i.e., species of algae that spend much of their life associated with the benthos) into the water column due to wind-induced resuspension. It is also important to distinguish between the terms "phytoplankton bloom" and "high primary productivity." High primary productivity refers to high rates of photosynthesis or growth. While phytoplankton blooms are generally associated with periods of elevated primary productivity, high primary productivity does not always result in high biomass due to the influence of loss processes (e.g., grazing, dilution, sedimentation).

The requisites for the formation of phytoplankton blooms include all factors that affect "gain" and "loss" processes, since blooms can only develop when biomass gains outweigh losses. The main "gain" processes are photosynthesis and growth which require the presence of adequate amounts of essential resources, including photosynthetically active radiation (i.e., PAR) and nutrients (i.e., carbon, phosphorus, nitrogen, other macro- and micronutrients, and in the case of species like diatoms silica) (Phlips, 2002; Reynolds, 2006). "Loss" processes include respiration, release of fixed carbon from cells, grazing, dilution/export of biomass, sedimentation of cells outside the euphotic zone, and cell death (i.e., programmed and disease related). Other physical and chemical factors play a modulating role in "gain" and "loss" processes, either by modifying the availability of growth resources or impacting the ecophysiology of the organisms themselves. For example, pH affects the availability of inorganic carbon and temperature regulates metabolic activity.

Historically, one of the most common uses of the term phytoplankton bloom has been in relation to the phenomenon of "spring blooms," which refer to peaks in phytoplankton biomass associated with the annual arrival in the spring of optimal light and nutrient conditions for phytoplankton growth in the surface mixed layer of many temperate marine ecosystems. Over the past few decades, research on phytoplankton blooms has increasingly focused on harmful algal blooms (i.e., HABs) and the impacts of anthropogenic sources of nutrients. Although phytoplankton blooms are a feature of many marine ecosystems around the world, even independent of human influences, intense accumulations of biomass of certain species can have disruptive consequences (i.e., "harmful" effects) for ecosystem structure and function, such as the production of toxins, depression of oxygen levels, severe light attenuation, alterations in food web structure, and related impacts on aquatic animal and human health (Moore et al., 2008). Concerns about the impacts of harmful algae blooms have grown with increasing evidence that human activities have increased the frequency, intensity, and magnitude of blooms (Heisler et al., 2008). Among these concerns are the potential effects of future changes in climate on phytoplankton blooms (Moore et al., 2008; Paerl and Huisman, 2009; Wetz and Yoskowitz, 2013).

Conclusions

The rapidly increasing body of knowledge on the ecophysiology of phytoplankton blooms and the growing availability of long-term data sets are providing a better picture of the factors that control blooms. These advancements, along with breakthroughs in computing and monitoring technologies, are providing the framework for the development of predictive models and effective management strategies.

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Cross-references

Algal Blooms
Anoxia, Hypoxia, and Dead Zones
Anthropogenic Impacts
Climate Change
Cyanobacteria
Eutrophication
Food Web/Trophic Dynamics
Macroalgal Blooms
Toxic Blooms

PNEUMATOPHORES

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Definition

Pneumatophores are specialized aerial roots stemming from a subterranean root system, which enable plants to utilize air in waterlogged soil habitats.

Description

Mangrove pneumatophores vary in size and morphological characteristics. The black mangroves (*Avicennia* spp.) have pencillike or fingerlike pneumatophores, while those of *Sonneratia* species are conical. Pneumatophores are usually unbranched, but branching may occur when damaged. Typical pneumatophore height is generally less than 30 cm for *Avicennia*, less than 20 cm for *Laguncularia*, and from 30 to 3 m for *Sonneratia* species. Pneumatophore density is typically large; for example, a 2–3 m-tall *Avicennia marina* tree usually has more than 10,000 pneumatophores. In *Avicennia* and *Sonneratia* genera, the pneumatophores contain chlorophyll in the subsurface layers and conduct photosynthesis in chlorophyllous layers under cuticles (Hovenden and Allaway, 1994; Duke, 2006).

As a kind of aerial roots, pneumatophore function is closely correlated with structure. Their surface is usually covered with lenticels, which permit outside air to diffuse into the spongy aerenchyma tissue. The aerenchyma is not only the oxygen container but also the oxygen transport pathway. Air is transported by aerenchyma to other parts of the plant, especially the underground roots below the pneumatophores that are tidally inundated. The pneumatophores are also pathways for carbon dioxide. Thus, oxygen can be pumped into the pneumatophores, enabling some mangrove species to survive in waterlogged conditions, while carbon dioxide, as a by-product of respiration, can be detected outside the pneumatophores (Scholander et al., 1955; Tomlinson, 1986).

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Cross-references

Mangroves

POLYCYCLIC AROMATIC HYDROCARBONS

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Synonyms

PAHs; Polynuclear aromatic hydrocarbons; Poly-aromatic hydrocarbons

Definition

Polycyclic aromatic hydrocarbons (PAHs) are a group of widespread atmospheric and aquatic pollutants of major concern because of their potential carcinogenicity, mutagenicity, and teratogenicity to a broad spectrum of organisms (Kennish, 1997). These organic xenobiotics consist of hydrogen and carbon arranged in the form of two or more fused aromatic (benzene) rings in linear, angular, or cluster arrangements with unsubstituted groups possibly attached to one or more rings (Eisler, 1987).

Description

PAH compounds range from naphthalene (two rings) to coronene (seven rings); the low-molecular-weight PAH compounds containing two or three rings are noncarcinogenic but acutely toxic to many estuarine and marine organisms, whereas the high-molecular-weight PAH compounds containing four or more rings have greater carcinogenic potential but are less toxic. Anthracene, fluorene, naphthalene, and phenanthrene are examples of low-molecular-weight PAH compounds that tend to be toxic; benzo(a)pyrene, benzo(c)phenanthrene, dibenzo(a,i) pyrene, and 3-methylcholanthrene are examples of high-molecular-weight, carcinogenic compounds. While some PAH compounds are synthesized by bacteria, fungi, and plants or are derived from natural fires, marine seeps, and volcanic emissions, most PAHs released to the environment originate from human activities (Kennish, 1992).

Highest levels of PAHs typically occur in urbanized estuaries and nearby coastal marine waters in close proximity to highly populated metropolitan centers. PAH-contaminated sites are often problematic sources of PAHs to local waterways (Lundstedt et al., 2007). The pyrolysis of organic matter, notably fossil fuels, is a primary delivery system of PAH compounds to estuarine environments. Other important delivery pathways include sewage releases, storm sewer runoff, oil refinery wastewaters, and oil spills. Urban and agricultural runoff and municipal and industrial discharges, therefore, play a major role in transporting PAH compounds to estuarine and coastal marine systems (Kennish, 2001).

PAHs are hydrophobic. Because they readily sorb to particulate matter in the water column and bottom sediments, these compounds partition out of the water column onto suspended particulates and bottom sediments through time (McElroy et al., 1989; Khodadoust et al., 2005). They are assimilated by benthic organisms and then pass through estuarine food webs. However, there are different degrees of bioavailability of these contaminants, and there are variable capacities of the organisms to metabolize them. Hence, uptake and concentrations of PAHs in biotic tissue can be highly variable as well. The concentrations of individual PAH contaminants measured in aquatic organisms generally range from 0.01 to more than 5,000 μg kg^{-1} dry wt (Kennish, 1997).

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Cross-references

Anthropogenic Impacts
Nonpoint Source Pollution

PRECAUTIONARY PRINCIPLE

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Synonyms

Precautionary approach; Preventive principle

Definition

The precautionary principle is an action or policy that should not be implemented if, following the assessment of the available information, it may represent a threat of serious, irreversible, or unpredictable damage to the environment or human health.

Precautionary risk management

The precautionary principle is an approach to risk management developed under circumstances of scientific uncertainty and where the consequences of a certain action may represent a potentially severe or unpredictable risk. The precautionary principle arose initially in 1970s German environmental legislation as the "Vorsorgeprinzip," regarding acid rain and justification for energy tax policies (Boehmer-Christiansen, 1994); "Vorsorgep" means foresight or anticipation with overtones of deliberated planning (Whiteside, 2006). At a higher level, it was recognized in the World Charter for Nature, adopted by the United Nations General Assembly in 1982. It was subsequently incorporated into various international conventions (CEC, 2000). This principle was enshrined at the 1992 Rio Conference, during which the Rio Declaration was adopted, whose Principle 15 specifies: "in order to protect the environment, the precautionary approach shall be widely applied by States according to their capability. Where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation" (UN, 1992). Although without referring to the precautionary principle specifically, it is generally regarded as the point of reference for this principle in the area of sustainable development.

A major challenge is how and when to apply the precautionary principle since social actors may want to take advantage of a current lack of scientific evidence to promote their own interests, misusing this principle for demagogy or protectionism purposes (Gollier and Treich, 2008). To avoid arbitrary decisions, the precautionary principle should only be invoked when three conditions are met: (1) identification of potentially adverse effects, (2) evaluation of the scientific data available, and (3) the extent of scientific uncertainty (CEC, 2000). If the principle is invoked, the following points should be considered: (1) measures adopted must be proportionate; (2) any decisions taken must be well founded, intelligible, and communicated; (3) measures adopted are provisional, and, if proven to be unsuitable considering new scientific findings, they should be adapted; and (4) an evaluation of the consequences of inaction (SFOPH, 2003). Decision makers, responsible for risk management, may then decide how to act, depending on the level of risk.

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Cross-references

Adaptive Management
Ecosystem-Based Management
Water Quality

PREDATOR–PREY RELATIONSHIPS

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Synonyms

Trophic interactions

Definition

A predator is an organism that eats another organism. The prey is the organism that the predator eats.

Introduction

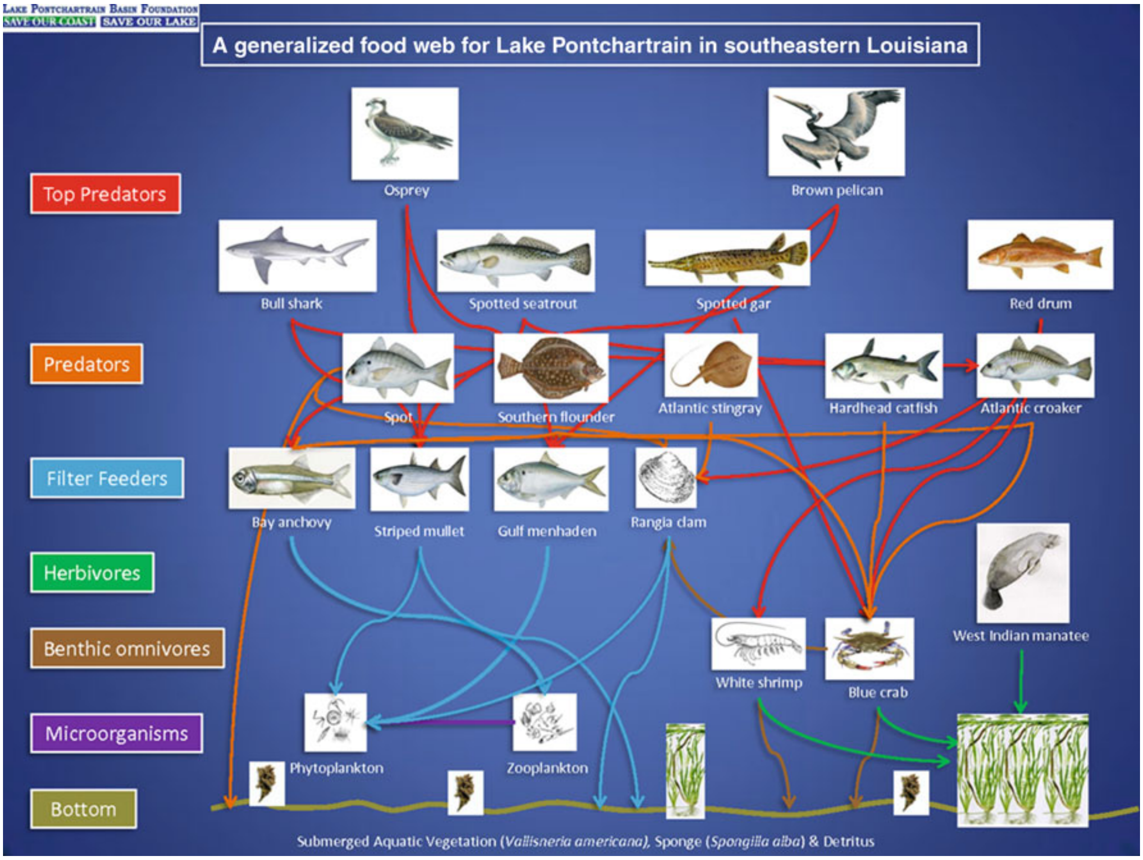
Intake of nutrients, i.e., eating, is essential in the long run; avoiding being eaten is even more important in the short term. Selective pressures often lead to an evolutionary "arms race" between prey and predator, resulting in improved prey capture by predators and antipredator adaptations by prey species.

Trophic levels

The trophic level of an organism is the position it occupies in a food chain. A food chain represents a succession of organisms that eat another organism and are, in turn, eaten. Food chains start at the first trophic level with primary producers (plants), followed by herbivores at the second level, predators at the third level, and typically finish with top predators at level 4 or 5. The number of steps that an organism is from the start of the chain is a measure of its trophic level. The path along the chain can form either a one-way flow or a more ecologically realistic food "web" in which a particular species eats and is eaten by a number of different species (Figure 1). The amount of energy available decreases by about a factor of 10 at each step.

Prey capture techniques and behaviors

Aquatic animals can acquire their food in a number of ways: Filter feeders or suspension feeders strain suspended matter and plankton from the water using a specialized filtering structure. Animals that use this method of feeding include attached and sedentary



Predator–Prey Relationships, Figure 1 Figure courtesy of Save Our Lake, Lake Pontchartrain Basin Foundation.

organisms, which generally cannot catch food any other way: bivalve mollusks, barnacles, corals, sea squirts, and sponges. Oysters can filter huge amounts of water and play an important role in clarifying water. Some mobile animals are also filter feeders, such as baleen whales and some fish such as menhaden.

Deposit feeders eat organic material and microalgae in sediments and include many benthic organisms such as polychaete worms, sea cucumbers, sea urchins, many snails, and some crabs. Their activities contribute to decomposition and nutrient cycles and circulate the sediments by bringing deeper sediments up to the surface. Neither of these two types of feeding represents true predation with predatory behavior.

Predators – active predators may hunt actively for prey. Some predators kill large prey and dismember or chew it prior to eating it, others may eat their (smaller) prey whole, as does a bottlenose dolphin swallowing a fish. Sea stars have an unusual style of predation – using steady strong pressure, they pry open the two shells of bivalve mollusks and then consume the animal inside.

Ambush predators capture or trap prey by stealth rather than by speed or strength. They usually hide motionless and wait for prey to come within striking distance. They often are camouflaged and may be solitary. Often, this type of predator cannot move faster than its preferred prey, so ambushing its prey is more effective than pursuit. Angler fishes (so named because they “go fishing”) are highly specialized ambush predators that have a “lure” on their head that resembles a worm or a small fish that they wave around to lure small fish into catching distance. The lure is a long, modified dorsal fin spine sprouting from the middle of its head that ends in a fleshy growth that can move and wiggle to resemble another animal. In some deep-sea anglerfish, this lure can emit light (bioluminescence).

Predator avoidance techniques and behaviors

Predator avoidance can be divided into two aspects, the first of which is not being noticed in the first place. This has both behavioral and morphological components. In order to effectively avoid and respond to a predator, animals must first identify the presence of a potential predator. The ability to recognize predator cues is essential for the initiation of antipredator behavior. Prey can rely on a variety of senses to detect predator cues, including visual, chemical, auditory, and tactile senses. Many prey species reduce their activity to prevent detection. This behavior can be stimulated in response to odors of a predator or alarm substances produced by conspecifics. Other behavioral responses include hiding (e.g., burrowing by fiddler crabs or burying in the bottom by flatfish). Many species of fish form schools of large numbers moving together. This safety-in-numbers approach has been found to be protective.

Many species utilize camouflage as protection. Many fish use countershading or counterillumination to reduce their visibility from both above and below. Some fish resemble a rock or seaweed, and decorator crabs put pieces of algae on top of their shells.

The mimic octopus can rapidly change its pattern and color to resemble a seaweed, fish, or rock and thus provide an excellent defense.

If the animal has been detected by a predator, there are additional techniques that may be employed, most obviously running or swimming away. Many animals have highly developed senses of sight, smell, and hearing so that they can detect danger and escape. However, escape has rather low odds of being successful, since the predator is probably larger and faster. This tactic is useful, however, when the predator is not too close and/or there is a refuge nearby. Another technique is to form a partnership (symbiosis) by joining with another species that is well defended. For example, some crabs carry sea anemones or sea urchins around, and clown fish ("Nemo") associate with sea anemones. Once an animal has failed at avoiding a predator or warding it off at a distance, self-defense may be utilized. Crabs and lobsters can use their claws to fight off predators. Puffer fish swell up and become larger when threatened. Morphological adaptations to reduce predation include having spines (e.g., sea urchins) that deter a predator or having a toxin, e.g., jellyfish and sea anemones with stinging cells on their tentacles. Some animals make use of signals, for example, bright warning coloration, advertising that they are poisonous.

If the animal has already been captured by a predator, there may still be some ways to survive. Mechanical defenses, such as armor and spines, can prevent access to softer edible parts. Many crustaceans have a reflex called "autotomy." A crab being picked up by a bird, for example, can break off its legs at a preformed breakage plane, thus leaving the predator with only a leg or claw while the crab scuttles away on its remaining legs. It can subsequently regenerate the lost appendages. Sea cucumbers under threat can eviscerate, ejecting part or all of their digestive tract at the predator.

Evolutionary aspects and plasticity

Predation can be an important evolutionary force since natural selection favors more effective predators and more evasive prey. "Arms races" have been recorded in some species. These need not require genetic selection over many generations, but can occur within an individual animal. These changes can be behavioral or morphological and are referred to as phenotypic plasticity. For example, periwinkle snails (*Littorina littorea*) in the presence of cues from their predators (green crabs *Carcinus maenas*) develop thicker shells, while the green crabs develop thicker claws with greater crushing power (Trussell and Smith, 2000; Smith, 2004). Mussels (*Mytilus edulis*) at sites with high levels of predation by green crabs developed thicker shells and greater shell mass and were more tightly attached (with byssus threads) to the substrate than mussels at sites with low predation. When mussels from low predation sites were moved to high predation sites, they produced more byssus threads and attached more firmly to the substrate. In the laboratory, mussels exposed to waterborne cues (odors) of the green crab grew thicker shells that were mechanically stronger and could better resist crushing (Leonard et al., 1999). However, mussels are also prey to sea stars, which use a totally different mode of attack. Instead of crushing the shell, these predators pry the two valves apart, using steady pressure. In order to defend against this type of predator, the bivalve needs to strengthen its adductor muscle, rather than thicken its shell (Freeman, 2007). Thus, induced defenses in mussels are specific to predators with different predation strategies.

"Bottom-up" and "top-down" effects

Populations are regulated by the amount of available nutrients. Thus, when nutrients (or prey) are high, populations of predators can thrive and grow. This is called "bottom-up regulation." However, populations can also be regulated by their predators, which is termed "top-down" control. Predation is a mechanism of population regulation. When the number of predators is scarce, the numbers of the prey species should rise because fewer of them are getting eaten. When there are more prey individuals around, the predators will have more to eat and would then be able to reproduce more and increase their population size. As the predator species increases, the numbers of prey decline. This results in food scarcity for predators, which will again have a decline in their population. Predation can have profound and complex effects on communities. The sea star, *Pisaster*, is the top predator in intertidal rocks in the Pacific Northwest. The rest of the community includes mollusks, barnacles, and other invertebrates. When sea stars were removed from an experimental area (Paine, 1974), barnacles and mussels began to occupy most of the available space on the rocks, outcompeting other species. The sea star was a keystone predator in this community, keeping the strongest competitors (barnacles and mussels) in check. Although it was a predator, the sea star helped to maintain a greater number of species in the community. Its beneficial impact on species that would otherwise be outcompeted is an example of an indirect effect.

A common scenario for observing food web effects is the invasion of a new species into an ecosystem. When nonnative species (exotics) invade an area, they often create "domino" effects, causing many other species to increase or decrease. Grosholz (2005) noted that as the green crab (*Carcinus maenas*) invasion in California progressed, populations of the native bivalve prey, *Nutricola*, decreased dramatically. At the same time, a previously introduced but rare bivalve (*Gemma gemma*) underwent a population explosion and spread, an example of what he termed "invasional meltdown."

Trophic cascades

Trophic cascades occur when predators in a food web suppress the abundance and/or alter traits (e.g., behavior) of their prey, thereby releasing the lower trophic level from predation (discussed above), and then this change affects the next lowest trophic level. For example, if the abundance of piscivorous fish increases, the abundance of their prey, zooplanktivorous fish, should decrease. This should be followed by an increase in abundance of large zooplankton since their predators have become scarce, and the large numbers of zooplankton should cause phytoplankton to decrease. A cascade in southern US salt marshes was noted by Silliman and Bertness (2002) who found that blue crab (*Callinectes sapidus*) predation on periwinkle snails (*Littorina saxatilis*) regulated the snail population size. Without the blue crab predators, the populations of periwinkles increased so much that their grazing on marsh grass could destroy the salt marsh, converting it into a mudflat. This trophic cascade suggested that overharvesting of blue crabs could be an important factor contributing to the die-off of salt marshes across the southeastern United States. Another example of a cascade is a study by Nifong and Silliman (2013) who focused on the next trophic level – the role of alligators (top predators) in southern US salt marshes. In mesocosms (large experimental tanks with many species present), they found that alligators reduced the abundance of blue crabs and induced behavioral changes, causing the crabs to decrease their own foraging activity and spend more time in a refuge. The effects of reduced crab abundance and reduced feeding behavior translated into increased survival of snails and mussels within the marsh mesocosms. Trophic cascades may also be important for understanding the effects of removing top predators from food webs, as humans have done in many places through fishing activities.

Summary

Predator–prey relationships are among the most important ones in ecosystems. They play major roles in structuring ecosystems and involve behavior, morphology, and evolution. They can be studied by manipulating ecosystems or observing changes in populations of native species following an invasion of a new species into an ecosystem.

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PUBLIC TRUST RIGHTS

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Definition

Coastal *public trust rights* include an indeterminate and evolving set of faculties grounded in the principle of free access to the coast. Primary coastal public trust rights that include pedestrian access, navigation, swimming, hunting, and fishing arose under the Roman Digest and have remained ever since. After recreational uses adjoined these customary faculties in the second half of the twentieth century, the latest coastal acts bear yet a third generation that incorporates education and research. According to the North Carolina General Statutes (1982), public trust rights are "those rights held in trust by the State for the use and benefit of the people of the State in common. They are established by common law [and] include, but are not limited to, the right to navigate, swim, hunt, fish, and enjoy all recreational activities in the watercourses of the State and the right to freely use and enjoy the State's ocean and estuarine beaches and public access to the beaches."

Defense of public trust rights

The defense of public uses constitutes one major objective in *coastal law*. For such purposes, conventional and national instruments provide many different devices, including specific servitudes granted to individuals or whole landlocked States (U.N., 1982) as well as special tools for land acquisition, or even the abstract characterization of the coast as *public domain* (Spanish Coastal Act).

Competing coastal uses

Coastal areas have long been the target of often-conflicting needs of human communities. In an attempt to settle competing demands, some coastal rights have become the object of reinforced protection, giving rise to the distinction between *exclusionary* and *public* uses. While exclusionary uses are those that limit or impede activities from others, public uses might be widely defined as rights equally and freely recognized by all citizens insofar as they do not hinder the rights of others. In turn, amidst public uses, those characterized by particular circumstances of risk or intensity are called *special*, in contrast to *general* ones (Parada, 2007).

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REDOX CONDITIONS

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Synonyms

Oxic-anoxic conditions; Oxidation-reduction conditions

Definition

Redox conditions refer to media dominated by oxidants (oxidizers, oxidizing agents, electron acceptors), the substances that have ability to oxidize other substances (cause them to lose electrons), or reductants (reducers, reducing agents, electron donors), the substances that have ability to reduce other substances (cause them to gain electrons).

Description

The content of dissolved oxygen (DO), the quintessential oxidizer in the natural waters, is used to define the following redox conditions:

Normoxic or *oxic* conditions correspond to high concentrations of oxygen (greater than 80 % saturation).

Oxygen deficient is a term used to characterize waters with oxygen concentrations below 80 % saturation or approximately 3 mg O_2 L^{-1} (about 95 µM O_2) (Renaud, 1986).

Hypoxic conditions correspond to the threshold of tolerance, stress responses, and morbidity of pelagic and benthic animals to low dissolved oxygen levels, usually in the range of 1–4 mg O_2 L^{-1} (e.g., Diaz, 2001; Vaquer-Sunyer and Duarte, 2008; Savchuk, 2010). A formal boundary is set at 2 mg O_2 L^{-1} (~63 µM O_2, i.e., CENR, 2000), or 2 ml O_2 L^{-1} (~89 µM O_2, i.e., Diaz and Rosenberg, 2008; Savchuk, 2010), or ~75 µM O_2 (Yakushev and Newton, 2013).

Suboxic conditions correspond to DO levels ~15 µM O_2, after which DO becomes an auxiliary oxidant, and the dominant electron acceptors are the oxidized ions of N (nitrate, nitrite) or oxidized species of metals: Mn(IV), Mn(III), and Fe(III). In the Black Sea, the reported suboxic values range from 4.5 µM O_2 (Lam et al., 2007) to 10 µM (Murray, 1991) and to 15 µM O_2 (Zubkov et al., 1992). Suboxic conditions may be further divided into *suboxidized* conditions, where DO is present in nanomolar concentrations, and *subreduced* conditions, where DO is absent (Stunzhas, 2005). The subreduced conditions, where there is no longer any DO and H_2S is not yet present, should correspond to the onset of processes such as anammox (Kuypers et al., 2003) or formation of Mn(III) (Trouwborst et al., 2006), which are inhibited by both DO and H_2S.

Once the oxidized species of nitrogen, manganese, or iron are completely depleted, OM microbial decomposition uses sulfate as the next available electron acceptor. This is the appropriate threshold for the term anoxic, where H_2S appears and the Eh potential becomes negative.

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Cross-references

Anoxia, Hypoxia, and Dead Zones
Dissolved Oxygen
Oxygen Depletion
Water Quality

RESIDENCE TIME

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Definition

Residence time is defined as (1) the time taken by a water parcel or particle to leave a defined region of interest (Zimmerman, 1976; Delhez et al., 2004; de Brauwere et al., 2011) or (2) the time for the mass of a scalar (or the volume of freshwater; Dyer, 1997; Geyer, 1997) within a water body to be renewed, given a known renewal rate (Bolin and Rodhe, 1973; Geyer et al., 2000).

Essential concepts, applications, and methods of estimation

"Residence time" is a commonly used term to generally convey time spent by water, dissolved substances, or particles within a defined region or water body. However, specific definitions and mathematical descriptions of this transport timescale vary considerably in the literature, as pointed out by others (Takeoka, 1984; Monsen et al., 2002; Sheldon and Alber, 2002). The definitions provided above are two of the most common and differ significantly from each other in their underlying assumptions, precise meanings, and methods of estimation. Care is needed in the selection and description of residence time and other transport timescale definitions and calculation methods, as estimated values and suitability for the scientific question at hand can vary substantially between approaches (Bolin and Rodhe, 1973; Monsen et al., 2002).

Following from the first definition above (i.e., the time taken by a water parcel or particle to leave a defined domain), residence time is a function of space and time of release, since particles originating at different locations and times within a water body require different amounts of time to leave (Monsen et al., 2002; Delhez et al., 2004). Defined in this way, residence time (time for a parcel originating at location **x** to exit a domain) is the complement of "age" (time the parcel took to reach **x** after entering the domain); the sum of the two timescales is called "transit time" (Takeoka, 1984; Sheldon and Alber, 2002). In tidal systems, ambiguity in the application of this definition may arise due to the fact that fluid elements may leave the region of interest on one phase of the tide and then reenter when the tide reverses. Therefore, many authors have employed a strict interpretation of this definition: the time required for a parcel to leave the domain *for the first time* (Monsen et al., 2002; Delhez et al., 2004; de Brauwere et al., 2011). Because the cumulative amount of time spent within a water body (over successive tidal visits) may be of ecological interest, a variation on the residence time concept – "exposure time" – has been defined and computed with numerical models (e.g., Monsen et al., 2002; de Brauwere et al., 2011). A variety of numerical approaches is employed to assess this first type of residence time, including Lagrangian particle tracking (Monsen et al., 2002), forward Eulerian (de Brauwere et al., 2011), and time-reversed Eulerian (Delhez et al., 2004) methods, the latter approaches making use of numerical tracers. Models representing a broad range of complexity have been implemented for this purpose, including box (Zimmerman, 1976), one-dimensional (Vallino and Hopkinson, 1998), two-dimensional (de Brauwere et al., 2011), and three-dimensional (Shen and Haas, 2004) models.

The second residence time definition (i.e., the time for scalar renewal) represents an integrative timescale describing general exchange characteristics of water bodies without addressing spatial variability (Monsen et al., 2002). Under steady-state conditions, residence time defined in this way takes the simple form M/F, where M is total scalar mass in a domain and F is the scalar rate of renewal (Bolin and Rodhe, 1973; Geyer et al., 2000). If freshwater renewal is of particular interest, freshwater volume may substitute for M and freshwater flow rate for F (Dyer, 1997; Geyer, 1997). These integrative timescales are also known as "flushing times" (Monsen et al., 2002; Sheldon and Alber, 2002) or "turnover times" (Bolin and Rodhe, 1973; Sheldon and Alber, 2006).

Regardless of definition, residence time is frequently compared with biogeochemical timescales to evaluate the relative importance of physical, biological, and chemical processes in aquatic systems (Vallino and Hopkinson, 1998; Dame, 2012). Residence time can thus be used, for example, to understand the fate of pollutants (Jonkers et al., 2005), the modulation of algal growth and loss processes in regulating primary productivity (Lucas and Thompson, 2012), and nutrient export from estuaries to the continental shelf (Nixon et al., 1996). In estuaries, residence time may be influenced by a range of physical forcings including river flow (Shen and Haas, 2004), tides (de Brye et al., 2012), and wind stress (Geyer, 1997). The relative importance of drivers may vary between estuaries (Geyer et al., 2000), spatially within estuaries (Shen and Haas, 2004), and over time (de Brauwere et al., 2011). Although different transport mechanisms can result in similar residence times, their effects on water quality and ecosystem function may not be equivalent (Geyer et al., 2000).

Summary

Residence time is one of several "transport timescales" that can be estimated to distill hydrodynamic processes to better understand their importance relative to biological and chemical processes and their influence on water quality and ecosystem function. Multiple specific definitions of residence time may be found in the literature, all meant to generally convey time spent by water (and the particles and substances carried with it) inside a defined region. Two of the most commonly employed definitions have been described herein: (1) the time for a water parcel or particle to exit a defined water body and (2) the time for the mass of a scalar or freshwater volume within a water body to be renewed. Methods for estimating residence times are highly varied, including field-based (e.g., Geyer, 1997) and numerical (e.g., Delhez et al., 2004) approaches.

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Cross-references

Age
Timescale

RESIDUAL CIRCULATION

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Definition

In estuaries *residual circulation* is defined as the circulation left after removal of the oscillatory tidal component from the current observations. In practice, this residual may also include contributions from "rectified" tidal propagations (the nonlinear interaction of tidal motion and bathymetry), direct (localized) wind forcing, indirect (large-scale) wind forcing, surface waves, and horizontal and vertical density gradients (Prandle, 2009).

Measurements and analysis

Because water motions occur on a continuum time of scale, it is critical to choose the appropriate time duration over which to estimate residual circulations (Kjerfve, 1989). In this case, the *residual circulation* will depend on the averaging time scale used. In physical oceanography analysis, there are several mathematical filters that can be used to separate the nontidal currents (due to wind forcing) into specific frequency bands. In estuaries, as most of the current variability usually occurs with a tidal periodicity, the residual circulation is usually calculated as the residual water movement after the currents is averaged over one, two, or numerous complete tidal cycles (Kjerfve, 1979).

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REVETMENTS

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Synonyms

Retaining wall

Definition

Revetments are a cover or facing of erosion-resistant material placed directly on an existing slope, bank, scarp, embankment, or dike to protect the adjacent shoreline and upland areas against scour caused by waves and currents (CHL, 2013).

Revetments consist of an armor layer, filter, and toe protection, and they depend on the underlying soil for support. Armor stone (high-energy environments) or riprap stone (lower-energy environments) must resist waves and settlement and provide drainage to ensure stability. The filter layer (e.g., smaller stone and geotextile fabrics) supports the armor stone and prevents retained soil from eroding through the armor layer by waves or groundwater seepage. Toe protection prevents scour and displacement of the seaward edge of the revetment. Revetments may increase erosion immediately seaward and adjacent to the structure (flanking) due to wave reflection while offering no protection to adjacent areas (USACE, 1981). Revetments should be built high enough or with an apron to prevent erosion at the top caused by wave run-up and overtopping.

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Cross-references

Bulkheads

RHODOLITH/MAERL

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Synonyms

Boxwork rhodolith; Coralline algal nodules; Maerl; Nucleated rhodoliths; Oncolith/oncoid; Prâlines; Rhodoid

Definition

Rhodolith/maerl is the most common term used for free living coralline red algae which live on sediments (Steller et al., 2003).

Description

The term rhodolith literally means “red stone” (Bosence, 1983a) and consists of the ancient Greek words for roselike (ῥόδειος) and stone (λίθοϚ). A rhodolith could be monospecific or multispecific, that is, if it consists of one or more than one coralline algal species. A rhodolith is developed around a nucleus of a rock, coral, shell, or any other element which serves as substrate for their development, but they also are found without any material in the center, usually consisting of a number of concentric *thallus* layers. Rhodoliths have a wide variety of growth forms ranging from fruticose to lumpy, and they are discoid or irregular shaped averaging approximately 6 cm in diameter (Foster, 2001). Rhodoliths are slow growing and can be long lived (>100 years), being resilient to varying environmental disturbances (Bosence, 1983b). They form a high-Mg calcite skeleton with periodic growth bands. Recent geochemical studies on coralline algae have focused on sea-surface temperature (SST) secular changes and have revealed a strong relationship with the variations of the Mg/Ca ratio in the algal skeleton at a monthly or biweekly resolution (Kamenos et al., 2008, 2009). They can form extensive beds, typically 30–100 % cover or more with layers over the seafloor, mostly in coarse clean gravels and sands or muddy mixed sediments. They can occur either on the open coast or in tide-swept channels of marine inlets. Rhodolith beds represent the transition between rocky and sandy areas (Foster, 2001) in which they may be mixed with kelp/fucalean forest or with seagrasses, but they can also cover extensive areas alone.

Rhodolith/maerl beds have been found worldwide, from low to high latitudes, at different depths, ranging from the intertidal down to 286 m (see review in Foster, 2001). Rhodoliths provide a stable, three-dimensional habitat onto which a wide variety of species can attach, including other algae, commercial species such as clams and scallops, and true corals. Although rhodoliths are resilient to a variety of environmental disturbances, they can be severely impacted by harvesting of commercial species (Steller et al., 2003), ocean acidification, or global warming (Martin and Gattuso, 2009).

Rhodoliths are a common feature of modern and ancient carbonate shelves worldwide (Nelson, 2009). Rhodolith communities contribute significantly to the global calcium carbonate budget (Amado-Filho et al., 2012), and fossil rhodoliths are commonly used to obtain paleoecologic and paleoclimatic information (Kamenos et al., 2008; Kamenos et al., 2009). More recently, the geological nomenclature indicates that the fossil counterparts of living rhodolith beds include a suite of terms such as rhodolith facies, coralline algal facies, rhodolith rudstone and floatstone, oncolithic rudstone, rhodolith pavements (= densely packed rhodolith rudstone), coralline algal nodule limestone, maerl pavement, red algal clast rudstone, and coralline branch rudstone and floatstone. Under the right conditions, rhodoliths can be the main carbonate sediment producers, often forming extensive deposits of beaches or sand dunes. They have an excellent fossil record since the Early Cretaceous (Aguirre et al., 2000).

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Cross-references

Macroalgae

RIA

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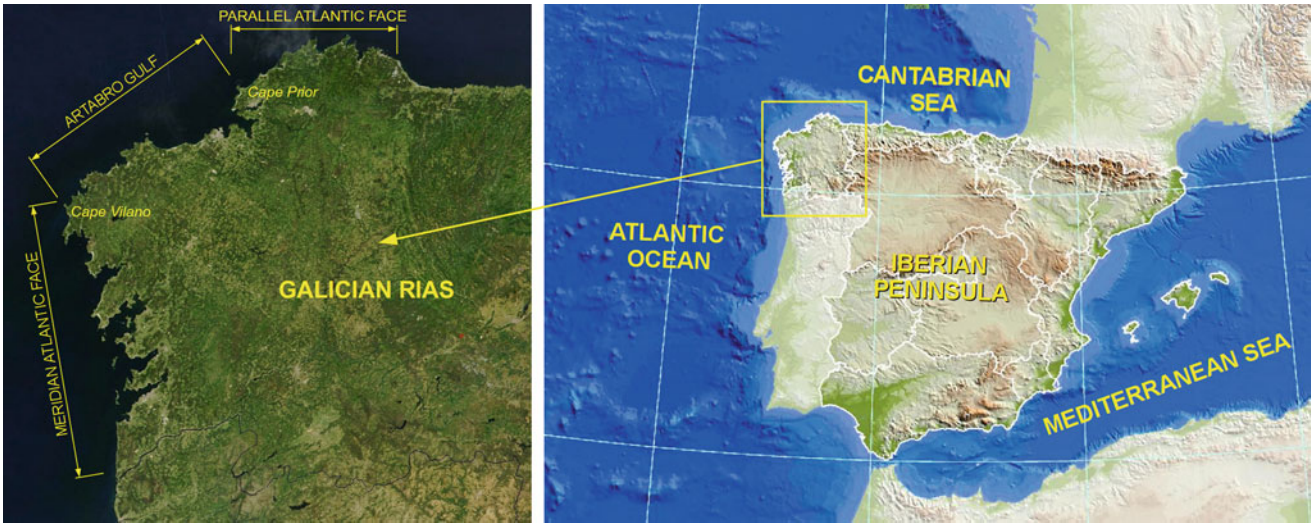
Synonyms

Flooded river valley



Ria, Figure 1 Rias in Spanish Galician and Cantabrian coasts. Font: NASA (*left*) and Spanish IGN (*right*).

Definition

Ria is a term used in Spain to refer to estuaries with a notable tectonic nature that are subjected to significant tides. Marine water predominates in these systems, and the mixture interface is highly vertical. They are common in Spain along the Cantabrian coast and in Galicia (Figure 1). A ria is a flooded valley like a fjord although fluvial in nature. They have many similarities with firths in terms of current infilling processes (McManus et al., 1993).

Genesis

Intrusion of the sea into fluvial valleys stems from two factors: Quaternary eustasy and tectonic sinking of the grabens on the fractured Atlantic edges of the Iberian plate. On the Atlantic side, this sinking has been continuous, but on the Cantabrian side, it has alternated with periods of uplift due to compression with the European plate. Its configuration corresponds to the current interglacial form, although the effects and relicts from previous periods are still evident.

There are several types of rias. In Galicia, they are differentiated by their tectonic history (Diez, 1996). Rias Bajas on the meridian Atlantic face, Rias Altas on the parallel Atlantic face, and Rias Artabras on the Artabro Gulf along a SW–NE diagonal. All are multilobed, which is a result of the successive graben blocks that were affected. Silting processes involving cohesive fluvial and sandy marine sediments (Asensio, 1979) dominate the most inland areas. The Rias Bajas are very wide and deep and follow SW–NE fracture lines. The Rias Altas are smaller and shallower but more perpendicular due to the effects of the European plate and have crossed fractures. The Rias Artabras are intermediate and converge at the center of the gulf.

The Cantabrian rias are located on a sea coast flattened by higher sea levels during the Quaternary period (*c. rasa*). Fracturing is much greater, and the mountain range is very close, which means that the river network is still poorly organized. This has resulted in rias that are much smaller and shallower than those in Galicia. There is a high degree of silting, and many beaches are created from marine sediment.

Other rias on the Iberian Peninsula (Aveiro and Formosa on the Portuguese coast and Tinto/Odiel in the Gulf of Cadiz) have more typical estuarine characteristics. The same applies to other rias around the world like those in Deseado (Patagonia) and Bahia Blanca and Tuyu (Buenos Aires) in Argentina. The mouth of the Georges River in Sydney, Australia, with its multiple lobes (lakes), and San Francisco Bay and Willapa Bay on the Pacific coast of the USA are also similar.

The rias of Lagartos and Celestún, Yucatan, Mexico, are further examples of the ocean penetrating inland, although they are of karstic origin. These circular arrangements of sinkholes (cenotes), which are also present on the Florida coast, are visible where marine sediment has yet to conceal the karstic landscape. This is perhaps why the much deeper karstic–fluvial coastal formations in Croatia are also known as rias. However, the case in Croatia must also be interpreted in light of the tectonic structure along the Adriatic coast. While they do not reach the size of those in Croatia, these formations are common throughout the calcareous Mediterranean. One example is the so-called *calas* on Spain's mainland and island coasts along the Mediterranean.

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Cross-references

Coastal Bays
Firth
Fjord

RIVER-DOMINATED ESTUARY

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Definition

A river-dominated estuary is one where fluvial processes extend to the estuary mouth where only ephemeral marine deposits occur.

Description

River-dominated estuaries tend to be shallow and undergo a cycle of infill related to floods from the catchments in their hinterland. During a non-flood period, a barrier is present at the seaward edge of the estuary backed by small flood-tide deltas where cohesive muddy and relatively steep channel banks are found. During a flood, the channels erode and the flood-tide deltas and barrier are often removed (Cooper et al., 1990; Cooper, 1993; Cooper, 2001). River-dominated estuaries where the barrier is formed of gravel-size sediment are also known as hapua (Hart, 2007; Kirk and Lauder, 2000). River-dominated estuaries often occur on high-energy, wave-dominated coasts, so they are unlikely to develop into a delta (Cooper, 1993).

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Cross-references

Tectonic Eustasy
Well-Mixed Estuary

ROCKY INTERTIDAL SHORES

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Definition

An intertidal rocky shore is the area between low tide and the highest reach of wave action on a rocky shoreline.

Introduction

Intertidal rocky shores in estuaries are a subset of all intertidal rocky shores. They are occupied by the same types of biota, which react to environmental factors and interact with each other in similar ways, although features of estuaries may impose modifications on these interactions. So, when discussing intertidal rocky shores in estuaries, it is necessary to consider general features of these habitats (Stephenson and Stephenson, 1972; Menge and Branch, 2001).

These habitats are occupied by a wide range of plants and animals. The plants are predominantly algae, including large seaweeds and microscopic forms. The animals are generally invertebrates, other than fish in pools, birds, and mammals which forage on shores. The invertebrates are very diverse at all taxonomic levels. In addition to mobile animals that include herbivores, predators, and scavengers, many animals are sessile filter feeders. What makes rocky shores different from many other habitats is the diversity of sessile animals ("ecosystem engineers," Jones et al., 1994) that create new, unique habitats. So, in addition to seaweeds which provide habitat for smaller species, animals, such as mussels, polychaetes, and ascidians, create large patches or reefs of unique habitat, which are then occupied by a range of other fauna.

Distributions and abundances of species on rocky shores

Although patterns, such as vertical zonation, are commonly described in the literature for rocky shores (Barnes and Hughes, 1999), quantitative data have shown that consistent broad-scale patterns of distribution are shown for only a few species and in some places. Despite strong gradients of submersion and emersion associated with the tide, there are no clearly defined heights on the shore where one assemblage of species abruptly changes into another. Instead, although different species occupy different heights, the change from the bottom to the top of the shore is a continuum of changes in species, so that the assemblage changes gradually.

More importantly, on most shores, even within their distributions, organisms show patchy abundances at scales of centimeters to tens of meters (Underwood and Chapman, 1996). This is due to ecological responses of species: (1) to abiotic factors, e.g., local topography or wave action; (2) to stochastic variation in processes such as recruitment or disturbances; and (3) to ecological interactions among individual organisms, e.g., competition, predation, or grazing. Thus, intertidal shores tend to be composed of patches of different sizes and ages, containing a suite of different species occurring in different abundances.

Distributions along estuaries

In the past, it was suggested that estuarine species are a discrete group, derived from marine and freshwater habitats. More recently, analyses show that, instead, estuarine assemblages are a continuum of marine and freshwater species that show gradual changes along the length of an estuary (Attrill and Rundle, 2002). Freshwater species become less abundant toward the mouth of an estuary, and marine species show the opposite pattern. There is a gradual changeover of species along an estuary, but no distinct estuarine assemblage with definite and recognizable boundaries.

Mobile species can change their position in an estuary as conditions change, e.g., with tidal penetration or seasonal changes in freshwater input. Species living on rocky shores tend to be attached, slow moving, or unable to migrate as adults from one shore to another. They must tolerate the range of environmental conditions to which any shore is subjected. Most species on rocky shores in estuaries are marine forms and must be able to survive reduced and widely fluctuating salinities and little wave action. They are a subset of the species that can live on open coast rocky shores, so diversity generally decreases inland; fewer marine species survive the environmental conditions or do not manage to recruit to sites up an estuary.

Processes affecting distribution and abundance

Wave action

On open coasts, waves are important in determining which species can survive on a shore. Areas with frequent large waves are only occupied by species that resist dislodgement. Shores with little wave action can be very stressful environments during low tide, depending on the local climate, because of extended periods of emersion without wetting by waves. Thus, low-shore species can extend often further up shore on wave-exposed shores, as long as wave action is not too severe. Because rocky shores in most estuaries are generally sheltered, except perhaps at the mouth of the estuary, waves probably have less influence in determining intertidal assemblages than they do on the open coast. In urbanized estuaries, large waves created by boats and ships can, however, replace the lack of natural waves (Bishop and Chapman, 2004).

Physical features of the shore

Physiological stress associated with high temperatures and desiccation during low tide is strongly modified by the shape and bedrock of the shore. Steep shores may be less affected by heat during low tide than are near-horizontal shores because steep shores are likely to be shaded for part of the day. Animals and plants may therefore be able to extend farther up shore, but this is countered by increased vulnerability to dislodgement by waves on steep shores. Species living on steep shores tend to have strong powers of adhesion or attachment.

On gently sloping shores, the type of bedrock becomes important. For example, sandstone does not get as hot as some other rock types, such as siltstone or basalt. It also tends to retain water during low tide. So, in estuaries where wave action is minimal, more species can be found on sandstone shores than on basalt shores, and they may be able to occupy a greater area on sandstone shores.

Intertidal distributions are also strongly influenced by local features of topography (Underwood and Chapman, 2007). Pits, crevices, the undersurfaces of boulders, and rock pools can provide shaded or damp conditions during emersion. They are often crowded with animals on estuarine shores during low tides on hot days, while the rest of the shore looks quite barren. Although the animals emerge from these habitats to feed during more benign conditions, many are slow moving and do not forage over large areas. Thus, the patchy nature of various microhabitats contributes to the patchy patterns of distribution of the animals sheltering in them and those animals and plants on which they feed.

Disturbances

Disturbances affecting intertidal assemblages may be widespread, e.g., flooding, waterborne pollution, or storms, or very localized, such as people trampling over the shore. They tend to damage and remove particular species from the assemblage, often those that are easily dislodged or broken or those particularly vulnerable to particular contaminants. Disturbances leave patches of bare space in reduced assemblages, space that is then available for other organisms to colonize. Recovery is, however, not predictable, and a different set of species may arrive in a disturbed patch to replace those lost. The new colonists may arrive as propagules from the water column, or as adults from surrounding areas. What arrives to colonize a patch will depend on larvae in the water column. It also depends on the variety of species living in the surrounding area and the rates at which they locate and encroach on the new patches.

Recruitment

Several processes remove the link between reproduction of adults and the young that arrive on the shore where they live. The majority of species release their propagules into the water column, where they are moved by waves and currents in the plankton, which may be a long period. During this time, they grow and metamorphose into larvae competent to settle, but are subjected to extreme levels of predation, which is very patchy. Thus, in a body of water that arrives on a shore, there may be many or few larvae, but the chances of larvae arriving on their shore of origin, when they are ready to settle and there is available habitat, are remote. Apart from species that are direct developers, e.g., some mollusks or some algae which drop their spores very close to the parent plant, there is little connection between breeding adult populations on a shore and new arrivals.

The importance of this invisible larval stage in determining future intertidal populations has been dubbed "supply-side ecology" (Underwood and Fairweather, 1989). It can result in very little recruitment on a shore, sometimes for many years, or the arrival of very large numbers of individuals, which can dominate a lot of the space. When there are large numbers of recruits and few resident predators, the recruits can survive, grow, and persist for many years. If there are few recruits and/or predators are very abundant, they will be quickly consumed and the space becomes available once again.

Many settling larvae have complex behaviors, which allow them to evaluate habitat before settlement. They may settle in response to features of habitat, such as small pits, or in response to cues from adults or other larvae. Together with patchy arrival in the water column, these behaviors increase local patchiness of species on a shore, especially for larvae that show gregarious behavior and settle predominantly where there are already adults.

Grazing and predation

Patterns of distribution and abundance of intertidal organisms are also strongly influenced by ecological interactions among species. Two of the most important of these are grazing and predation.

Many grazers, such as amphipods and micro-gastropods, are small and tend to live in cryptic habitats, such as on seaweeds. These species have little impact on the distributions of species on a shore, but affect the epibiota growing on the larger plants. The major grazers on rocky shores are mollusks, particularly limpets. These have major influences on the distribution of seaweeds because they consume the microalgal stages of development. These grazers are very competitive, and food is generally in short supply; therefore, they tend to consume most of microscopic stages of the algae before they have had the chance to grow. So, large algae and animals and plants, which use them as habitat, tend to be confined to areas without large numbers of grazers. This is generally lower on the shore, where algae can grow fast and occupy a lot of space, excluding large grazers (Underwood and Chapman, 2007).

The most important predators on rocky shores are starfish, crabs, and whelks, although predatory starfish are usually not common on sheltered shores. Crabs are common predators and scavengers on rocky shores, particularly low shore, and in boulder fields where they can shelter under stones. They feed primarily on slow-moving gastropods, which they crush or "peel" to remove the shell. The common European crab, *Carcinus maenas*, can extend far into estuaries, although it dies in freshwater.

Other common predators on rocky shores are whelks. These are slow moving and feed primarily on sessile animals, such as barnacles and mussels. They may take many hours, or even days, to eat a single item of prey, so they often do not feed far from areas of shelter. This behavior can cause haloes of bare space around shelters, when prey near the shelter have been consumed and the survivors are too far away to be preyed upon. This increases patchiness created by variable patterns of recruitment or survival in response to physical stress.

In contrast to open coasts, fish in estuaries can be important predators on intertidal shores, especially in very sheltered areas. For example, the common toadfish, *Tetractenos hamiltoni*, frequents shallow water in estuaries in New South Wales, Australia. Although generally eating benthic prey in soft sediments, they can remove all the small oysters from patches of shore.

Indirect interactions

It is inevitable that within a limited number of pages, one can only describe some of the major and most important processes affecting intertidal assemblages. With such a diverse biota, there are also many indirect interactions (Wootton, 1993). For example, if a species (A) feeds on one (B) of two species which compete, large numbers of the predator may promote the survival of the competitor which is not consumed (C), by removing its competitor, even though there is no direct interaction between A and C. Removal of mussels by whelks might thus promote survival of barnacles, which provide habitat for other species. Similarly, where large grazers cannot feed in very complex topography or among patches of sessile animals such as barnacles or mussels, the algae can escape grazing and grow to form patches outside their general range. Thus, the more diverse the assemblage, the more possible for its species to interact in direct and indirect ways and the less likely that the interactions will be similar from place to place and their outcomes in terms of spatial patterns of distribution and abundances predictable (Underwood and Chapman, 2007).

Temporal processes acting on rocky shores

Most of this discussion has focused on spatial patterns of distributions and abundances. Although it is not possible to go into detail here, processes alter with time, depending on rates and types of arrivals onto a shore, regular (e.g., seasonal) and irregular (e.g., storms) disturbances, and changes in the intertidal assemblage itself. Thus, consumption of barnacles by whelks may lead to patches being dominated by mussels until storms remove patches of mussels, when barnacles, more mussels, or grazers may appear in the patches. The spatial structure of an intertidal assemblage is not static, but dynamic, and this, too, interacts at many spatial and temporal scales (Dayton, 1971).

Other hard intertidal habitats in estuaries

In addition to rocky shores, there are other intertidal habitats composed of hard surfaces, i.e., not beaches, mudflats, or mangroves, in estuaries.

Boulder fields

Boulder fields are important intertidal habitats for a wide range of species. In addition to their upper surfaces, which support similar assemblages to those on rocky shores, their undersurfaces provide a unique sheltered habitat. This is occupied by species that are vulnerable to harsh environmental conditions, which affect intertidal assemblages during low tides. Thus, they often support similar species to those found in rock pools.

On open coasts, boulders are vulnerable to overturning by waves, which tends to reduce the number of species living on and under them. Boulder fields in estuaries have more sheltered conditions and can support diverse assemblages. In areas that are urbanized or popular for recreation, human disturbances may be very large, e.g., overturning boulders while foraging for food or bait. They may also be subject to burial or being smothered by silt, particularly when rivers flood. How damaging this is to the animals and plants depends on how long the disturbance lasts before normal conditions reappear.

Built habitats

With increasing coastal urbanization, a common feature of many estuaries is an ever-increasing expanse of artificial shoreline (Dugan et al., 2011). This includes walls to prevent erosion, marinas and wharves for shipping and boating, pylons of bridges and aquaculture facilities, among others. They create novel intertidal "rocky shores," although they may be made of a mix of different materials, natural and manmade. In some areas, they impinge on or have replaced soft sedimentary habitat. In others, they have replaced natural rocky shores. In some part of the world, they are the only intertidal hard substrata available in very urbanized estuaries.

These artificial shores can have major impacts on native intertidal biodiversity. First, they may provide hard substrata where it does not naturally occur, allowing species to move into new areas, potentially influencing the native biota. Even when replacing natural rocky shores, artificial surfaces do not act as surrogates of natural shores and may not support the same diversity of intertidal biota, even when created from natural local rock. Rarer species seem most vulnerable to loss when shores are replaced by seawalls as many of the common species can live on both types of structure. So, although many artificial structures seem to support a thriving and diverse intertidal assemblage, one cannot be complacent that these structures can act as surrogates of natural shores (Chapman and Underwood, 2011).

Summary

Estuarine rocky intertidal habitats are like similar habitats on open coasts, which are sheltered from large waves, with some modifications. Estuaries generally have irregular pulses of freshwater, which reduces the number of species that live on estuarine rocky shores. Nevertheless, they are often occupied by a suite of micro- and macroalgae, sessile habitat-forming animals, herbivores, predators, and scavengers. Interactions among these species and their responses to local conditions are complex, occur at multiple spatial scales, and vary through time. This variability, natural to all intertidal assemblages, makes their responses to climatic change and other anthropogenic disturbances difficult to predict. Yet with increasing urbanization in and around estuarine areas, these assemblages are becoming more vulnerable to disturbances from which they may not fully recover.

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Cross-references

Emergent Shoreline
Estuarine Habitat Restoration
Herbivorous Grazers
Intertidal Zonation
Littoral Zone
Macroalgae
Predator–Prey Relationships

ROCKY SHORE

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Definition

A rocky shore is a coastline that is dominated by outcrops of consolidated geology. All rock types can form a rocky shore. The essential difference between hard (rocky) and soft shores is whether rocks or unconsolidated sediment forms the primary landforms.

Description

Rocky coasts are erosional environments and are a source of sediment for the surrounding landscape. There are three broad types of rocky shores: plunging cliffs, type A shore platforms, and type B shore platforms (Sunamura, 1992). Plunging cliffs are vertical or semi-vertical faces where their base occurs in deep water. Shore platforms are ledges that occur at the base of cliffs and either form a seaward sloping ramp (type A) or are semi-horizontal with a distinct break in slope and vertical face at their seaward edge (type B). Rocky shore evolution is driven by both marine and subaerial processes (Kennedy et al., 2011). Where the former dominates, cliffs tend to be vertical in profile while the latter produces more convex profiles (Emery and Kuhn, 1982). The evolution of rocky shores is primarily related to the erodability of the rock structure in which they are cut (Stephenson et al., 2013), and as a result in harder lithologies, rocky shore morphology may evolve over multiple eustatic sea-level cycles (Trenhaile, 1987; Trenhaile, 2001).

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Cross-references

Emergent Shoreline
Shoreline
Shoreline Changes
Submergent Shoreline

S

SALT MARSH ACCRETION

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Definition

Salt marsh accretion is defined as growth by deposition of suspended particles during flooding (allochthonous growth) and by accumulation of plant material, both roots and decomposed material from plants growing in the marsh (autochthonous growth) (Schuerch et al., 2012).

Process

Salt marsh accretion consists of the addition of both sediment and detritus from above and adding root tissue below the salt marsh surface. The sediment components, both organic and inorganic suspended solids, are waterborne particles that have been trapped in the salt marsh vegetation. The detritus is decaying plant material from dieback following the growing season. Added together, the peaty soil of a salt marsh was reported to be >80 % organic matter in a Chesapeake Bay salt marsh (Langley et al., 2009) and, in a Rhode Island salt marsh, 91 % and 96 % in low and high marsh, respectively (Bricker-Urso et al., 1989). There are many factors influencing the rate and quality of accretion, as shown in Figure 1. The result of the many interacting factors is that salt marsh accretion is keeping ahead of sea-level rise in many but not all regions.

Suspended solids

Materials washing downstream or resuspended from under open water tend to settle out in salt marshes, where the water movement is slow. These materials include both organic and inorganic components. Since suspended sediment particles settle slowly to the bottom, settling out is aided by natural meandering streams; conversely, if straight channels are dug into the marsh to aid drainage (as occurs with ditching for mosquito “control”), then the tide ebbs faster with less time for settling out. If the upstream materials are prevented from reaching an estuary, as occurs with dams on and dredging of rivers, and by construction (urban development), a marsh has less material to accumulate for the accretion process.

Vegetation

Since high and low marsh have different species of vegetation and since different species are characteristic of these marsh segments on the several continents, it follows that if the several species cause different rates of accretion to occur, then different marsh areas and geographic regions will have different accretion rates. The phenomena relating to accretion include stem density, plant biomass (both above and below the marsh surface), and rate of detritus formation following autumnal senescence. In fact, it has been suggested, on the basis of both observations and modeling, that vegetation actively engineers its landscape, building its environment to soil heights that optimize its stability (Marani et al., 2013).

Stem density

The flow of water during tidal flooding is somewhat impeded by the plant stems. The physical transport of suspended solids is thus slowed down – reduced 16 % according to Wolanski et al. (2000) – allowing more time for (1) adhering to plant stems and (2) settling out of the water column to the marsh surface.

When observing a salt marsh at low tide, the high tide line on plant stems can be seen because sediments have adhered to the intertidal portion of the vegetation. However, Li and Yang (2009) quantified the amount of sediment *adhering to* vs. *trapped by* three species of plants

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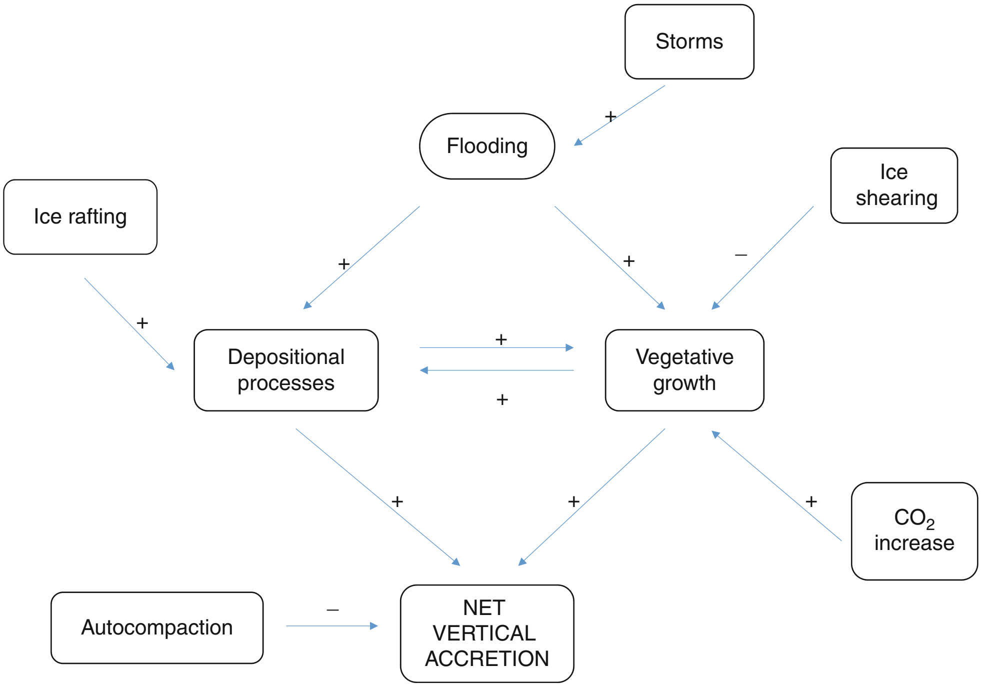

Salt Marsh Accretion, Figure 1 Modified from Chmura et al., 2001.

in the Yangtze River Delta. They found that 5–10 times as much weight of sediment settled into joints of leaves and stems than adhered directly to the leaves and stems; the total amount of biomass correlated directly with the amount of adherent sediment; the amount of adherent sediment decreased with elevation from the marsh surface and with the distance from the marsh edge toward the high marsh (both of these obviously reflect the amount of time of water contact during tidal ebb and flow); and the invasive (in that region) species *Spartina alterniflora* trapped significantly more sediment than the native *Phragmites alterniflora* and *Scirpus mariqueter*. All of these plant species die back in the fall, adding their sediment-laden detritus to the marsh surface.

CO_2 increase

As atmospheric CO_2 increases, its effect on stimulating plant growth has been studied in many areas. With salt marsh vegetation in particular, a mesocosm experiment in which CO_2 was nearly doubled (simulating expectations for the year 2100) enhanced growth, thickening the soil by 4.9 mm yr^{-1} vs. the control plot increase of 0.7 mm yr^{-1}, in part due to stimulation of fine root growth (Langley et al., 2009). This effect is greater in C_3 plants such as the sedge *Schoenoplectus americanus* than in C_4 plants such as the grasses *Spartina* spp. and *Distichlis spicata* (Cherry et al., 2009; Langley et al., 2009). In the latter, it was not just the CO_2 fertilization effect that was observed but also an interaction between CO_2 and salinity (amelioration of salt stress).

Flooding/Sea-level rise

In a greenhouse experiment designed to replicate another result of climate change, flooding stimulated root growth (Nyman et al. 2006), thus adding to the organic portion of the marsh soil.

Ice

When wind-driven ice flows are marooned on a marsh, the debris in the ice can add to the marsh surface. On a marsh in Maine, this process added anywhere from 0 % to 100 % of the total surficial accumulation (Wood et al., 1989). This phenomenon is obviously more relevant at higher latitudes and thus with global warming can be expected to decrease in significance. On the other hand, ice shearing can be destructive, removing both organic and inorganic material from the marsh surface (see Figure 1).

Autocompaction

As marsh soils continue to accrete, the accumulated weight squeezes out the water from deeper layers, lowering the overall height of the marsh (Orson et al., 1998), thus decreasing the height of the apparent accretion.

Meteorological forcing

In low marsh areas, which normally have more time under water, *storm strength* was determined to be a major factor in enhancing accretion. In high marsh areas, which normally have less time under water, it is *storm frequency* that is relevant (Schuerch et al., 2012). This was determined with cores of marshes in Germany's Wadden Sea where geochronology was done by radioisotope measurements (see "Methods of measuring accretion", below) and comparing depths with hydrologic data going back to 1938.

Methods of measuring accretion

Vertical cores taken through salt marsh soils are sliced into disks. These disks can then be measured for the radioisotopes ^{137}Ce and ^{210}Pb. These radioisotopes fall to earth with precipitation. The ^{137}Ce was most heavily deposited from the atmosphere in 1963, the final year of atmospheric nuclear weapon testing, so the depth at which the greatest amount is found indicates when that level was deposited. On the other hand, ^{210}Pb forms steadily in the upper atmosphere as a result of interaction between atmospheric Pb and cosmic radiation; perturbations in the amount of this steady deposition can be correlated with historical knowledge of meteorological events, both wet and dry. The ^{137}Ce measurement is useful at the yearly level, while ^{210}Pb measurement is useful at the decadal level. The analysis of pollen grains in the soils is useful at the century level if local changes in plant cover are known, such as the conversion of forested land to agriculture.

Summary

As seen above, there are many factors that interact in the accretion process. CO_2 increase enhances plant growth, but more in C_3 salt marsh plants, which are less salt tolerant, than in C_4 salt marsh plants, which tend to be more salt tolerant. Flooding, on the other hand, as is occurring more often with sea-level rise, can bring higher salinity to a given marsh. In many areas, salt marsh accretion is keeping up with global climate change-induced sea-level rise. However, this resilience, as represented by stable equilibria, may be susceptible to "changes in the relative sea-level rise. . . with consequent reductions in the associated biodiversity" (Marani et al., 2013).

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Cross-references

Climate Change
Mosquito Ditching
Sediment Resuspension
Sediment Transport
Tides

SALTMARSHES

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Definition

The coastal saltmarsh ecosystem is defined by both habitat and biota. It is intertidal, occurring on soft shores, and the plant communities are comprised of herbaceous flowering plants, both monocotyledonous and dicotyledonous, and small shrubs. The ecosystem can be distinguished from mangroves which are dominated by trees and in which herbaceous understory is absent or rare (Janzen, 1985) and from sea grass communities which are predominantly subtidal or, if intertidal, occur lower on the shore than saltmarsh (see *Mangroves*).

Introduction

Saltmarshes are conspicuous features of many estuaries and are one of the most intensively studied ecosystems in the world (Chapman, 1960; Ranwell, 1972; Long and Mason, 1983; Adam, 1990; Pennings and Bertness, 2001; Perillo et al., 2009; Saintilan, 2009a) (see *Coastal Wetlands*).

Coastal saltmarsh is found on low-energy coasts, predominantly in estuaries, but also in the shelter of barriers or where wave energy is dissipated before reaching the marsh. Saltmarsh may be found on the shores of coastal lagoons, both permanently open lagoons and ICOLLs (Intermittently Closed and Open Lakes and Lagoons;

Haines, 2012). ICOLLs are of widespread occurrence, and some are very large. While lagoon ecosystems have many distinctive features (Barnes, 1980), the fringing saltmarshes are, at least floristically, little different from those found in permanent estuaries in the same region.

Saltmarshes occupy the interface between terrestrial and marine environments (Valiela et al., 2001) and have many features and attributes of both, as well as some unique features. The boundaries of saltmarsh may be abrupt and easily defined (Figure 1), or their recognition may require setting an arbitrary limit along a continuum of change (Pratolonga et al., 2009). Even if, to the human eye, the landward boundary is sharp, there may be fresh groundwater flowing under a marsh, affecting water relations of saltmarsh plants and determining biogeochemical processes (Valiela et al., 2001; Boorman, 2009). Valiela et al. (2001) suggest that more attention should be given to studying exchanges into, out of, and within marshes than to boundaries per se.

At their seaward edge, saltmarshes may abut sand/mudflats, sea grass beds, or, in tropical and subtropical regions, mangroves. At the landward side of saltmarshes, there are often artificial boundaries to embankments, built development, or transitions to shingle/cobble beaches (Bertness, 2007), sand dunes, or some form of swamp forest or freshwater wetland. The upper boundary of saltmarsh is frequently marked by the presence of a driftline of wrack and other material (increasingly plastic; Adam, 1990; Wolanski et al., 2009). The natural components of wrack may originate from within the marsh itself or be carried into the estuary from the catchment (timber, leaf litter) or from the sea (algae and sea grasses). Material may be deposited by high tides at different levels within the marsh during the course of the year (Beeftink, 1979). Some material may subsequently be carried back to the sea, but much will eventually accumulate at the level of the highest tides of the year. At localities close to human settlement, the wrack may be collected as fertilizer for local farms or gardens, but, increasingly, it is seen as unsightly and a source of odors and is removed to "tidy up" the coast. Nevertheless, wrack provides a significant habitat, and decomposition of wrack provides an important pathway for nutrient recycling.

Global variation

Saltmarshes occur at suitable locations on all continents except Antarctica. They are generally regarded as a feature of temperate coasts being replaced in the same positions in the landscape in the tropics and subtropics by mangroves (Wolanski et al., 2009). While mangroves are restricted to tropics and subtropics with only minor incursions into temperate zones (most extensively in Australasia), saltmarshes occur in the tropics in two situations: (1) on coasts with reliable rainfall, evenly distributed throughout the year where they may be found as a narrow fringe above mangroves, and (2) as much more extensive saltmarsh stands occurring on drier, often markedly seasonal, coasts where hypersalinity in the upper intertidal precludes mangrove development (Adam, 1990; Costa et al., 2009). As salinity increases, vascular plants in the upper intertidal become very sparsely distributed, although microalgal and microbial mats may be extensive; any boundary between flats and sparse saltmarsh is arbitrary.

Saltmarshes, Figure 1 Saltmarsh abutting the limestone cliff of Humphrey Head, Morecambe Bay, northwest England.

Saltmarshes as ecosystems support a wide range of organisms, but it is the vascular plants which are the most obvious component. The distribution of plant taxa permits the recognition of different types of saltmarsh at the global scale (Chapman, 1953; Chapman, 1960; Chapman, 1977). Chapman's classification was adopted by Ibanez et al. (2013). Adam (1990) proposed a variant of this model, suggesting broad similarities between temperate marshes in both hemispheres. With one conspicuous exception, *Spartina* dominated marshes on the West Atlantic coast of the Americas, the distribution of the major types of saltmarsh can be related to latitude. In terms of floristic richness, saltmarshes provide an exception to the general rule that within biomes, species richness is greatest in the tropics and declines with increasing latitude. Tropical saltmarshes have very low species diversity, and the highest diversity is shown in high latitude temperate marshes. This pattern is shown both in the northern hemisphere (Adam, 1990) and in Australia (Saintilan, 2009b; Boon et al., 2011) and South America (Issach et al., 2006), although in this latter case, at the highest latitudes in Tierra del Fuego, there are floristic links to northern hemisphere temperate marshes. Saltmarshes on coasts experiencing Mediterranean climates share strong similarities in floristics, vegetation structure, and physiognomy. Whether biogeographic patterns of saltmarsh fauna at a global scale are congruent with floristics has not been fully investigated.

Variation within estuaries

Environmental conditions vary in space and time within estuaries, and this is associated with changes in the fringing saltmarshes (Ranwell, 1968). Some changes relate to the stage of geomorphological evolution of estuaries (Roy, 1984; Roy et al., 2001), others to the gradient in salinity from the mouth to the head of the estuary. In general, conditions within saltmarshes are more saline towards the mouth of an estuary, while at the limits of incursion of saline water, freshwater tidal marshes may occur (Odum, 1988; Temmerman et al., 2003). The head of estuaries was often the location of the first established human settlements, so that there is a very long history of destruction of freshwater tidal marshes, which in many parts of the world are now rare. Between the saline and the freshwater zone, there may be brackish marshes, which are variable in their species composition (Sainty et al., 2012). Species characteristic of more saline sites may occur in the brackish zone, but the zonation patterns may be inverted (Gillham, 1957), with the saline zone species being restricted to the higher, less frequently flooded parts of marshes, where evapotranspiration between flooding tides results in more saline conditions.

Variation within marshes

One of the almost universal features of saltmarshes is the zonation of species from low marsh to the upper tidal limit (Figure 2). The proportion of the tidal range occupied by saltmarsh varies between sites; where *Spartina* spp. occupy the lower zone they are able to endure more frequent inundation than other species. In Victoria (Australia), the introduced *Spartina anglica* occurs below the mangrove zone (*Avicennia marina*) (Boon et al., 2011). The boundaries between zones are often (but not necessarily) sharp even if the topographic gradient is slight (Marani et al., 2013) (see *Species Zonation*).

Saltmarshes, Figure 2 Zonation of saltmarsh vegetation at Humphrey Head, Morecambe Bay, northwest England. Low marsh (to *left*) dominated by *Puccinellia maritima*, mid-marsh dominated by *Festuca rubra*, and upper marsh dominated by *Juncus maritimus*.

Two principal interpretations of zonations have been advanced (Davy, 2000): that they are a spatial expression change over time (succession) or that the distribution of species is controlled by environmental factors and/or competitive interactions between species. Davy (2000) warned against too ready assumption of successional interpretation and emphasized that succession and zonation should not be treated as synonymous concepts.

Static zonations are widespread; for example, on rocky intertidal shores or as the bands of vegetation related to altitude on mountains. The position of boundaries between species, or between communities, is determined by species' responses to environmental factors and by interactions between species. The patterns could be redrawn by climate change, major tectonic events, or the invasion of introduced species but, in the absence of these external forces, will remain stable. There are also many well-documented examples of zonations which reflect succession, for example, the plant communities on moraines deposited by retreating glaciers.

Are the zonations on saltmarshes expressions of succession? The zonation may continue beyond the tidal limit, for example, the zones of swamp forest (*Casuarina, Melaleuca*, and *Eucalyptus*) found inland of saltmarsh in eastern Australia (Pidgeon, 1940). While Pidgeon (1940) suggested that this zonation reflected succession, there is little evidence for progression from saltmarsh to nontidal communities. In many parts of the world, the most seaward saltmarsh communities today are dominated by introduced species of *Spartina*. The zonation of communities above the *Spartina* zone reflects succession from the original (pre-*Spartina*) pioneer, but, if the marshes continue to prograde and succession from *Spartina* occurs, it is unlikely that there will be a recapitulation of historic succession, rather a new range of communities may develop (Adam, 1990; Davy, 2000). During the course of marsh development, management regimes may change. The introduction or the removal of livestock, or changes in the abundance of native grazing animals, can change the occurrence and abundance of plant species, so that, even if zone boundaries remain at the same elevation, the species composition within zones may change, so the communities now occurring are not those present at the same elevations earlier in the marshes' development.

In northern temperate regions at the end of the last glaciations, the newly exposed land surface would have been prone to erosion, but sea level was much lower than at present so deposition would have occurred at locations now beneath the sea. When sea level reached its present position about 6000 years ago, the landscape was well vegetated and rates of erosion would have declined. Clearance of forests and the development of agriculture would have initiated a new phase of increased erosion. In Europe, this would have occurred more than 1,000 years ago, but in North America, major clearing following European colonization was only a few centuries ago (Kirwan et al., 2011). In North America, the increased sediment input into estuaries stimulated a phase of saltmarsh growth (Kirwan et al., 2011). Recognition of the damage caused by erosion in catchments and the initiation of soil conservation measures has reduced sediment input into rivers. Construction of dams and other water management works has reduced sediment transport into estuaries throughout the world (Walling, 2006; Walling, 2008). A consequence of this is that in a number of major estuary systems (e.g., Mississippi, Nile, Yangtze, and Venice lagoon), saltmarshes are now suffering "sediment starvation," compromising the ability of saltmarshes to respond to disturbance or to future rises in sea level. Mudd (2011) has suggested release of pulses of sediment into estuaries to preserve marshes.

Succession on saltmarshes

The development of saltmarshes is affected by a range of physical processes and interactions between these processes and the biota, both flora and fauna (Allen, 2000; Reed, 2000). The position of saltmarsh relates to that of sea level, which over geological time has varied considerably. Following the last glacial maximum, sea level rose to reach approximately its present position 6000 years ago. Since then, there have been considerable changes in the occurrence and extent of intertidal marshes. There is ample empirical evidence that saltmarshes can develop, or erode away, in short periods (Oliver, 1906; Oliver, 1907; Packham and Liddle, 1970; Pringle, 1995; Adam, 2000; Davy, 2000). As a marsh develops, it can both prograde (extend farther seaward) and accrete (increase in surface elevation). The two processes frequently co-occur, but accretion can be maintained, or even increased, if, when, a marsh front is eroding, sediment released by erosion is deposited on the remaining marsh.

Low- and mid-marsh zones can develop over short periods – decades (Adam, 2000) or one or two centuries (Pethick, 1980; Pethick, 1981). Upper marshes may be stable for very much longer – 2,000 + years (Pethick, 1980; Pethick, 1981) in East Anglia, United Kingdom, and 4,000 + years in northeastern United States (Redfield, 1972). Upper marsh zones are frequently not homogeneous, but are complex patterns of species and communities reflecting the operation of a range of ecological factors rather than simple succession.

Accretion requires an increase in surface height, generally as a result of accumulation of sediment, either minerogenic (allochthonous) – sand, silt, or clay carried in by the tide – or autochthonous accumulation of organic material (from plant roots or incorporated stems and leaves) or varying combinations of the two. Marsh growth commences with the establishment of pioneer plants, which promotes sedimentation. In the earlier stages of marsh development, sediment accumulation is not necessarily continuous in space or time. Sediment deposited may be eroded. However, as the density of vegetation increases, more sediment is retained, and the marsh surface rises. With increasing elevation, the rate of accretion will decline as the number of flooding tides decreases and the flooding water will have already passed through the vegetation of the lower marsh where sediment will have been deposited. The rate of allochthonous sedimentation would be expected to fall close to zero at the tidal limit. Studies in eastern England (The Wash – Kestner, 1975; North Norfolk – Pethick, 1981) showed that, while accretion in the lower marsh declined as elevation increased, as the model predicted, the upper marsh surface reached an asymptote lower than that of the highest predicted tide. Allochthonous sedimentation on vegetated surfaces occurs in two ways: through capture of sediment onto leaves and stems and through settling out of sediment from water stilled by the vegetation (Mudd et al., 2010). Input of sediment is not the only factor determining marsh elevation; Cahoon (2006) identified eight processes which influence changes in elevation, both positively and negatively. Sediment input could be countered by, for example, compaction or shrinkage (see *Salt Marsh Accretion*).

The dropping of sediment or accumulation of organic material creates a topographic gradient. The separation of species along this gradient to give rise to zonation is a response to a variety of factors. Conventional wisdom suggests that the lower (seaward) limit of species is determined by responses to physical factors, while interaction between species is more important in the upper marsh. However, even at the lowest limits, marsh biotic interactions can be important. Doubts that the lower limit of the pioneer *Salicornia europaea* in southeast England was set by tidal action were raised by Gerdol and Hughes (1993), who showed that the lower limit of *Salicornia* coincided with the upper limit of the amphipod *Corophium volutator.* Removal of *Corophium* permitted seedlings transplanted to lower levels to survive, suggesting that disturbance of the sediment by the amphipod prevented *Salicornia* from reaching its physically determined lower limit.

Creeks and pans

Saltmarshes are rarely simple inclined planes; generally, they have complex internal topographic variation, with creek and pan systems present. The form of creek systems varies considerably. On very sandy marshes, the creek systems are simple and less dense than those on muddy substrates (Chapman, 1960). Pye (2000) recognized six different arrangements of creek forms in southeast England.

Creeks, as well as providing drainage, represent an extension of the estuary water body and mudflat habitats into the marsh. Creeks also generate additional topographic complexity within marshes. As flooding tides overflow the creek banks, sediment is deposited to create levees and, consequently, basins between creeks (Beeftink, 1977; Temmerman et al., 2004). The well-drained soils of the levees provide a contrast to the poorly drained, frequently anoxic, conditions of the basins. This environmental difference is reflected either by the occurrence of different species assemblages on the levees or in the basins (Beeftink, 1977; Adam, 1990) or, in species poor zones, by taller growth on the levees.

As well as creek systems, the marsh surface may be interrupted by pans, steep-sided pools. The density of pans varies considerably between marshes and in some sites is low. In southeast Australia, where marshes occur as a zone on the landward side of mangroves, both creeks and pans are absent from many sites, although shallow bare areas may occur (Figure 4) (Adam, 1997). Pan development occurs early in succession, with patchy distribution of pioneer plants leading to sediment deposition around plants, and depressions between them (Yapp et al., 1917; Goudie, 2013) (Figure 3a). As the vegetated surface continues to rise and consolidate, the depressions form pans (Figure 3b). In addition, localized bank slumping produces blockages in creeks resulting in chains of channel pans (Yapp et al., 1917). If pans were formed only early in succession, then their density would be similar regardless of marsh age. Pethick (1974) demonstrated that in eastern England the density of pans increased in high mature marsh, so the Yapp et al. (1917) model is not a complete explanation of pan formation. There are a number of different possible mechanisms of formation of additional pans. Pethick (1974) suggested that local deposition of wrack could smother vegetation, leading to death followed by erosion to create pans. On some coasts, ice scour might also be a mechanism of pan formation in high marshes. Importantly, the upper marsh pans described by Pethick (1974) have the same general form as those discussed by Yapp et al. (1917) – steep sided and generally water filled (as illustrated by Figure 2.3 in Steers, 1977). Also widespread in some high-marsh zones are extensive shallow depressions, either bare or sparsely vegetated, which are variously referred to as pans, pannes, or, as in Clarke and Hannon (1967), "rotten spots." These might develop as a consequence of poor drainage (Ewanchuk and Bertness, 2004) or as a result of the development of a hypersalinity through evapotranspiration in summer. The extent of these bare areas may vary over time. Hamilton (1919) described more widespread and extensive bare areas in the marshes of the Sydney region (Australia) in the early twentieth century than are present in the early twenty-first century. Loss of vegetation may also be caused by the use of recreational vehicles (Kelleway, 2005), and naturally bare areas may be extended, or have recolonization prevented, by vehicle use (Figure 4).

The environment

Environmental conditions within saltmarshes are very much determined by the tide. The tidal regime varies between sites, and estuarine saltmarshes experience the full range of tidal regimes. Tidal ranges vary from macro, including locations with the largest ranges in the world (Bay of Fundy, the Bristol Channel), to negligible. In some regions, there is a seasonal variation in water level, for example, in the Baltic (Gillner, 1965), or the estuaries of southwest Western Australia (Brearley, 2005). ICOLLs may, when closed, be nontidal for extended periods, but, nevertheless, the conditions established during opening periods determine the structure and function of any fringing saltmarshes.

The number of tides reaching particular positions on the marsh surface varies with elevation. The lower marsh may

Saltmarshes, Figure 3 (**a**) Pioneer saltmarsh in the Duddon Estuary, northwest England. Sedimentation in patches of *Puccinellia maritima* creates hummocks, with bare areas in between. (**b**) Pans in a saltmarsh in the Leven estuary, northwest England. A heavily grazed marsh with the vegetation 2–5 cm tall.

be reached by nearly every tide (although where mangroves are found as a seaward zone even the lowest levels of marsh may be flooded by relatively few tides). The upper levels of marsh are reached by spring tides, at the highest levels possibly only once or twice a year. In the lower marsh, with frequent flooding, soil salinity and soil aeration are reasonably constant; at higher elevations, there is greater variability. Conditions are strongly influenced by climate. Rainfall can lower soil salinity, and dry periods can result in hypersalinity. The tidal range will determine the depth to which the lower marshes flood and importantly, the velocity of the tidal current. The speed of the current can influence the success of seedling establishment, newly germinated seedlings being vulnerable to uprooting. The timing of tides and of rainfall (germination of many halophytes is promoted by salinity reduction) means opportunities for seedling establishment are limited.

Explanations of the patterns of species and community distributions within saltmarshes have traditionally emphasized the importance of physical factors and assigned a limited role to biotic interactions, an example being the description of the holocoenotic complex by Clarke and Hannon (1969). Nevertheless, the role of plants was recognized very early in scientific investigations of saltmarsh geomorphology. Similarly, the role of grazing by livestock

Saltmarshes, Figure 4 A large bare area in a saltmarsh on the Kurnell Peninsula, Botany Bay, Australia. This panne may be of natural origin, but any regeneration has been limited by the use of the area by off-road bicycles.

in determining the species composition of vegetation has long been recognized in Europe (Adam, 1978). However, until recently, the importance of invertebrates and the effects of human activities in estuarine ecosystems were not fully appreciated (Silliman et al., 2009a). In the United States, there have now been several convincing demonstrations that key consumers can alter vegetation, for example, following increases in populations of snails (Silliman et al., 2009a) and crabs (Holdredge et al., 2008; Altieri et al., 2012). The increases in grazers were most probably initiated by reduction of predator numbers as a result of harvesting by both professional and recreational fishers.

Dieback

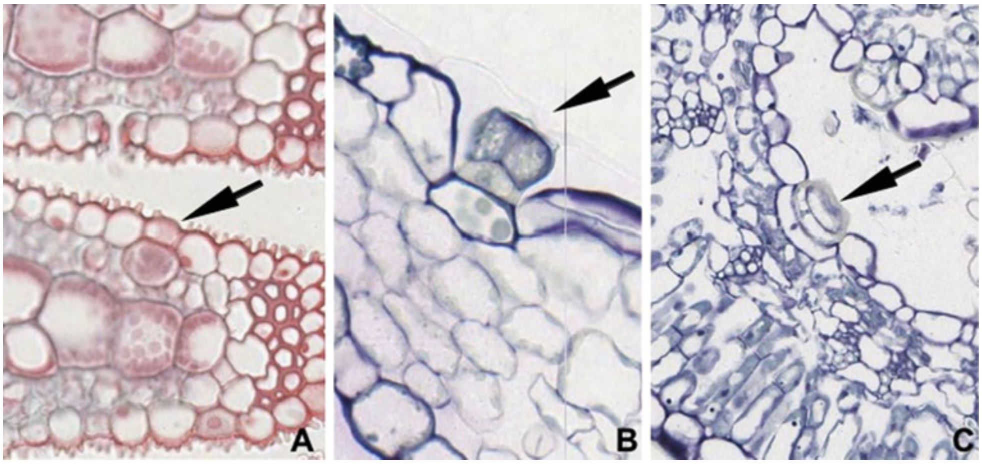
In the 1950s, extensive areas of *Spartina anglica* marshes in southern England exhibited dieback (Goodman et al., 1959), although despite considerable investigation, no single causal factor was isolated. In the early twenty-first century, dieback of thousands of hectares of *Spartina*-dominated marshes on the Gulf and Atlantic coasts of the United States also occurred (Alber et al., 2008; Osgood and Silliman, 2009). A single causative factor is unlikely and Osgood and Silliman (2009) suggest that interactions between several factors are involved: (1) climate, with severe drought being a trigger; (2) drought, which could be responsible for changes in soils, including increased salinity, acidity, and bioavailability of metals, the stresses either directly killing *Spartina* rendering it more susceptible to pathogens; and (3) populations of grazers which have been increased by reduction in predator pressure. Osgood and Silliman (2009) discount eutrophication as a major factor in dieback, although acknowledging that it contributes to the general stresses on marshes.

Eutrophication is widespread in estuarine and coastal waters in the United States (Scavia and Bricker, 2006) and elsewhere in the world. Deegan et al., (2012) propose that eutrophication is a driver of saltmarsh loss. The mechanism suggested is that nutrient enrichment promotes increased aboveground productivity and reduction (or maintenance of) below ground biomass so that there is a reduction in the binding of sediment, leading to collapse of creek sides and erosion of the lower marsh zone.

The mechanisms suggested as contributing to both dieback and losses from eutrophication are not unique to North America. Why have similar phenomena not been reported from saltmarshes in other continents? Are there features of *Spartina* marshes which render them particularly vulnerable?

Biota

The biota of saltmarshes can be categorized in a number of different ways – by conventional taxonomy, by whether species are of marine or terrestrial origin, and by pattern of occupancy of saltmarsh habitats (permanent residence, residence for particular stages of life cycle, migrant, or opportunistic).



Saltmarshes, Figure 5 A diversity of salt glands in saltmarsh plants. (**a**). *Spartina anglica.* (**b**). *Samolus repens*. (**c**). *Frankenia pauciflora.*

Vascular plants

The vascular plants growing in the intertidal zone face two particular challenges: salinity and the consequences of inundation. Of these, salinity has been the most studied, and there is now a detailed understanding of the mechanisms conferring salt tolerance (Flowers et al., 1977; Flowers et al., 1986; Flowers and Colmer, 2008; Flowers et al., 2010; Munns and Tester, 2008). Plants which tolerate saline conditions are referred to as halophytes, in contrast to non-salt-tolerant species, the glycophytes. Halophytes are found not only in intertidal saltmarshes but in a range of other salt-affected habitats. As yet there is not a complete list of halophytes, in part because there are parts of the world where saline habitats have not been fully investigated but also because of lack of agreement on the definition of halophyte. There is a continuum of salinity conditions and where within the transition from brackish to freshwater conditions a limit is drawn between the habitats of halophytes and glycophytes is arbitrary. Adopting the definition of ability to complete the life cycle in at least 200 mM salt (Flowers and Colmer, 2008; Flowers et al., 2010), the number of halophytes is probably between 500 and 1,000. In some cases, an entire taxon might be described as halophytic, but there are a number of widespread species where only some genotypes exhibit halophytic traits (Adam, 1990). Some of the species-rich upper saltmarsh communities in northwest Europe (Adam, 1981) may be composed, at the species level, of species also found in a range of nonsaline habitats. Nevertheless, the saltmarsh forms of the species are adapted to tolerate and thrive under moderately saline conditions and can thus be regarded as halophytic genotypes. This emphasizes the point made by Flowers et al. (2010) that halophytes are not unique; all plants possess the same features, but the pattern and degree of expression of particular traits are different in halophytes from those in glycophytes.

Halophytes include species from a number of different families and orders; the ability to tolerate saline conditions appears to have evolved independently on a number of occasions (Flowers et al., 2010). Viewed in isolation, there are no features which immediately identify a plant as being a saltmarsh species. This differs from the much smaller pool of species which constitute mangroves, where features such as possession of aerial roots or production of viviparous seedlings are characteristic. A feature shown by many saltmarsh species is succulence, but this is also displayed by species in a number of other habitats and on its own would not be sufficient to identify species as being from saltmarsh. One feature that might indicate that a species is a halophyte is the possession of salt-excreting glands (Figure 5) which can be detected by the occurrence of salt crystals on the surface of leaves. Salt glands are also found in a number of mangrove species. In terms of structure salt glands are similar to the range of other secretory structures found in plants.

Saltmarsh plants lack the obvious external root modifications displayed by mangroves, but like many other wetland plants possess abundant aerenchyma tissue (Armstrong 1978), so that aerobic conditions are maintained within tissues even when the surrounding sediment is anoxic.

Saltmarshes and mangroves today are overwhelmingly dominated by flowering plants. Nevertheless, the estuarine landforms they occupy existed on earth long before the evolution of the flowering plants some 65 million years ago. Did earlier land plants form saltmarshes and mangroves?

The fossil record of plants is patchy. Many do not have structures which are candidates for fossilization, and many land plant habitats are unlikely to be environments where in situ fossilization could occur. Nevertheless, the world's major coal deposits represent the former vegetation of vast wetlands, many of them occurring in deltas under brackish or saline conditions. Fossils in these deposits reveal the occurrence of large arborescent early vascular plants, which formed communities analogous to mangroves.

It is curious that today none of the survivors of those plant lineages occur in mangroves, with the exception of the fern *Acrostichum aureum*, which is widespread throughout the tropics (Tomlinson, 1986). There is no indication from the pre-flowering plant record of saltmarsh analogues. One of the earliest land plants was *Rhynia*, which occurred in what Channing and Edwards (2009) interpreted as a hot mineral-rich spring similar to those found at Yellowstone National Park today. Some at least of the earliest land plants therefore lived in harsh environments, and the high mineral content of the springs might suggest that tolerance of salinity would have been possible. However, there is no support from the fossil record for early occurrence of extensive saltmarsh.

Bryophytes are likely to have been early components of the terrestrial flora, but are not generally recognized as components of saltmarsh floras. However, a number of bryophyte species occur in tidal saltmarshes and in inland saline habitats (Adam, 1976; Adam, 1990; Garbary et al., 2008); therefore, while bryophytes are frequently absent and are rarely abundant in saltmarshes, they demonstrate a capacity to tolerate salinity and might have been more abundant in the past.

The present-day vascular flora of saltmarshes is diverse. Barbier et al. (2011) refer to saltmarshes as grasslands, but while large areas of the world's saltmarshes are grasslands or dominated by physiognomically similar graminoids (sedges and rushes), there are also large areas dominated by herbs or shrubs, particularly by genera of Amaranthaceae (Figure 6).

Most saltmarsh communities are dominated by perennial species. Annual plants are found in the low marsh zone in those parts of the world where *Salicornia* occurs and in microhabitats in the mid and upper marsh where disturbance creates openings in which annual plants are able to germinate.

Algae

Algae are important components of saltmarshes. Microalgae are not always apparent to the human eye, but microalgal films contribute to marsh formation and ecosystem productivity. Many microalgae have mucus coatings, and this is important in binding sediment (Coles, 1979; Underwood, 2000). Establishment of microalgae on mudflats may promote development of microtopography (van de Koppel et al., 2012), facilitate the establishment of pioneer vascular plants (Coles, 1979), and protect the surface of the lower marsh from erosion. Mason et al. (2003) showed that microalgae were susceptible to even low concentrations of triazine herbicides, suggesting that runoff of herbicides from agricultural land into estuaries might initiate erosion of saltmarshes.

Macroalgae may be conspicuous components of saltmarshes. In western Ireland and western Scotland, very dense swards of dwarf fucoids (only 1 or 2 cm tall) dominate lower marshes (Adam, 1981), and on Atlantic saltmarshes in North America and numerous localities in Europe, larger fucoids contribute substantially to marsh productivity (Roman et al., 1990) creating protective microclimates and providing habitat for intertidal invertebrates (Tyrrell et al., 2012). Eutrophication of waterways can promote overgrowth and smothering of saltmarshes by green macroalgae (McComb and Lukatelich, 1995), particularly by species of *Ulva*.

Microorganisms

Microorganisms play essential roles in ecosystem processes in saltmarshes, playing important roles in nitrogen, sulfur, and other biogeochemical cycles and in breakdown of detritus. Microbial films on detritus are an important food source for browsing invertebrates.

Recently, there has been considerable interest in the interactions between halophytes and microorganisms in the rhizosphere (Ruppel et al., 2013). Bacteria increasing the availability of phosphorus and iron have been isolated from the rhizosphere of several halophytes (Ruppel et al., 2013). Arbuscular mycorrhizae, which enhance nutrient uptake by plants, were first reported from the roots of halophytes early in the twentieth century (Adam, 1990; Davy et al., 2000), but only recently has there been investigation of their functions (Fuzy et al., 2008; Evelin et al., 2009). Studies so far have concentrated on individual species; placing mycorrhizae in the context of the functions of saltmarsh ecosystems is a research challenge for the future.

Fauna

The majority of animal phyla can be found in saltmarshes, although many groups have been little studied. A review of the fauna of saltmarshes was provided by Daiber (1982). The majority of the literature assessed in that study was from North America; the number of studies outside America has subsequently increased, but there are still extensive geographic regions and many faunal groups for which few data are available.

Marine fauna

Saltmarsh creeks are an extension of the estuarine water body, and for nekton (species within the water column) movement between the main estuary and creeks is readily possible. On high tides when creeks are bank full and overflow to flood the marsh surface, nekton can move from the creeks onto the marsh surface. There are, however, differences in the age and size structure of nekton assemblages in marshes compared with those in the main estuary. Saltmarsh creeks are important nursery habitats for many fish and crustacean species and are thus essential for supporting the estuary ecosystem. As a consequence, they also sustain many commercial fisheries upon which local communities may depend for employment and from which the larger human community gains part of its food supply.

Saltmarshes, Figure 6 Saltmarshes are not always grasslands. (**a**) Saltmarsh at Bosham, southern England. The large grey-foliaged shrub is *Atriplex (Halimione) portulacoides.* (**b**) Saltmarsh at the Sydney Olympic Park, Homebush Bay, Australia, dominated by the succulent subshrub *Sarcocornia quinqueflora* (Amaranthaceae (Chenopodiaceae)). The marsh is fringed by swamp oak (*Casuarina glauca*) forest.

Upper saltmarshes are visited by nekton when flooded by spring tides; these flooding events are brief and may not occur on every spring tide cycle during the year. As such, the utilization of the upper marsh by nekton has been considered to be accidental and probably of little value to the visitors. Studies in eastern Australia challenge this assumption (Hollingsworth and Connolly, 2006; Mazumder et al., 2006; Mazumder, 2009; Platell and Freewater, 2009; Mazumder et al., 2011). Breeding of saltmarsh crabs is linked to the tidal cycle, with release of zoea larvae coincident with high spring tides. Individual larvae are very small, but they are extremely abundant. They represent a high-quality food resource for fish, which consume them in large numbers. Thus, although high marshes are limited in extent and only accessible for short periods of time, they may play a disproportionately large role in estuarine fish ecology, both directly for species which access the marshes and indirectly higher up the food chain in estuaries and adjacent coastal waters.

The marine benthic fauna is diverse and occupies a range of habitats. The regularly inundated unvegetated banks and sides of creeks support a similar range of species to those of intertidal mudflats. Marsh surfaces are habitat for a range of species including large numbers of crabs and marine gastropod molluscs. Within the sediment, there is an abundance and diverse range of meiofauna, whose survival in anaerobic mud may be enabled by the creation of an oxygenated rhizosphere (only a few millimeters thick), sheathing the roots of vascular plants as a result of oxygen loss from aerenchyma tissues (Teal and Kenwisher, 1966; Teal and Wieser, 1966; Osenga and Coull, 1983). Interactions between larger benthic species and flora may be important. The sediment input into marshes may contain a high proportion of fecal and pseudofecal pellets (Frey and Basan, 1985), which, as well as influencing particle size, may be an important source of bioavailable nitrogen and phosphorus for plant growth (Long and Mason, 1983). Crab species may be present in large numbers and, through their burrowing, may influence microtopography, sediment aeration, and chemistry and drainage (Bertness and Miller, 1984).

The interactions between the fiddler crab *Uca pugnax* and the dominant vascular plant *Spartina alterniflora* were described by Bertness (1985). The crabs are absent from the mudflats in front of the marsh, as the sediment is insufficiently consolidated to permit burrows to be maintained. They are also absent from areas of the short form of *Spartina alterniflora* where the dense root map precludes burrow formation. However, they are abundant in stands of tall *Spartina alterniflora* where root densities are lower. Bertness (1985) proposed that there was a facultative mutualism between the crabs and tall *Spartina* such that the plant root density was sufficient to provide firm sediment to maintain burrows but not so dense as to prevent burrowing, while improved drainage and aeration created by burrows promotes plant productivity and maintains the tall growth form of the grass. Not all marshes support such abundant crab populations, but the example demonstrates the ability of invertebrates to structure the environment, and even in the absence of crabs, other taxa may play a similar, if quantitatively lesser, role.

The spatial scale over which individual invertebrates on saltmarshes feed may be small. Saintilan and Mazumder (2010) took advantage of the difference in photosynthetic pathway in the two dominants of the saltmarsh in eastern Australia, the C_4 grass *Sporobolus virginicus* and the C_3 succulent subshrub *Sarcocornia quinqueflora*, to investigate the feeding ecology of two grazing crab species and a marine gastropod. The different photosynthetic pathways result in the plant tissues having different carbon isotope signatures which can be traced from the plants to herbivores. The two plants occur in a mosaic of virtually monospecific patches. The study showed that *Sporobolus* was the species directly consumed and that individual grazers had territories of at most a few square meters. In the *Sarcocornia*-dominated patches, the major energy source for the fauna was fine organic material, rather than living plant tissues.

Terrestrial fauna

The terrestrially derived fauna of saltmarshes is large, but many components of it have not been as well studied as the marine component.

Invertebrates

For many members of the public, the terrestrial fauna which are of most concern are mosquitoes and biting insects. There are a number of diseases which can be transmitted to humans by mosquitoes, and with growing human populations close to saltmarshes and the possibility of increased insect populations and greater incidence of pathogens as a consequence of global warming, there is likely to be greater pressure for implementation of control measures (Dale and Breitfuss, 2009). There has been a long history of ditch construction in saltmarshes in the United States with the object of reducing insect populations (Gedan et al., 2009), but the consequences of the ditching for the whole ecosystem were not initially given consideration. In subtropical saltmarshes in Australia, shallow ditches, referred to as runnels, have been dug and have apparently been successful in reducing mosquito populations but with few other effects on the marshes (Dale and Breitfuss, 2009).

Many of the insects in saltmarshes do not encounter the impacts of tidal flooding, as they utilize tall vegetation which remains emergent even at high tide. However, herbivorous species (grazers and sap suckers) must be adapted to process plant tissues with high salt content and low water potentials.

There are other species which are behavioral and physical adaptations to either survive or avoid the impacts of tidal submergence (Treherne and Foster, 1979; Foster WA, 2000). Some of the most challenging conditions for insects on saltmarshes are experienced by species of aphids which live on plant roots (Foster WA, 2000).

Beetles can utilize saltmarsh pools, although not necessarily at all stages of their life cycle (Foster GN, 2000). There is also a diversity of ground-living beetles in upper marsh grasslands (Luff and Eyre, 2000). The food plants of the larvae of a number of larvae can survive occasional submergence by seawater (Agassiz, 2000). Some saltmarsh lepidopterans are rare and of conservation concern (Agassiz, 2000; Relf and New, 2009), but others are very abundant, for example, in Australia the small Samphire Blue *Theclinesthes sulpitius* whose larvae feed on samphires (succulent shrubby chenopods) (Orr and Kitching, 2010).

Anthills occur in other saltmarshes in northern Europe and, although higher than the surrounding marsh surface, may be completely submerged during spring tides (Kay and Woodell, 1976). Anthills provide a habitat for a number of plant species which are otherwise restricted in the distribution in marshes (Woodell, 1974). There are a number of reports of ants occurring in saltmarshes elsewhere. The role of ants in processes such as seed distribution and pollination in saltmarshes has not been investigated (Adam, 1990).

There is a considerable diversity of spiders in saltmarshes, and they are major predators in the ecosystem (Barnes, 1953; Heydemann, 1979; Grimshaw, 1982; Petillon et al., 2007). Some species escape flooding tides, but a number of species can survive inundation (Petillon et al., 2009).

Terrestrial invertebrates on saltmarshes exhibit a zonation comparable to that of plants (Heydemann, 1979). In some cases, this is due to a direct relationship with particular plant species, while in other cases, it is due to the animals responding to a similar environmental gradient which determines plant distribution.

Terrestrial vertebrates

The most obvious vertebrates utilizing saltmarshes are birds. Many species fly over saltmarshes or are occasional visitors, but there are species which are particularly associated with saltmarshes.

In the northern hemisphere, large flocks of waterfowl – ducks, geese, and swans – are a feature of many saltmarshes for at least part of the year. Many of the species are migratory, breeding in the Arctic in summer and overwintering farther south. The populations of a number of these species, particularly geese, have increased considerably in recent decades primarily in response to changed agricultural practices which have increased the availability of winter food, leading to higher survival rates. In North America, the lesser snow goose (*Chen caerulescens caerulescens*) population breeding adjacent to Hudson Bay in northern Canada has caused extensive damage to coastal marshes (Jefferies et al., 2006), with loss of vegetation results from the grubbing up of roots and rhizomes; the development of hypersalinity in the newly bare areas and the compaction of the soil through trampling by numerous geese prevent recolonization (Jefferies et al., 2006; Henry and Jefferies, 2009). On the wintering grounds, grubbing by geese can be a major but localized disturbance (Smith and Odum, 1981), but geese also utilize extensive areas of cropland, so the damage to temperate marshes is far less than that in the Arctic. In northern Europe, overwintering waterfowl help to maintain low grass vegetation on saltmarshes. Whether impacts of increased breeding populations on Eurasian Arctic marshes are as extensive as those in North America is not reported in readily accessible literature. The impact of geese is a very striking example of consumer pressure controlling ecosystem structure.

Shorebirds (which include wading birds) also include many migratory species. The migratory paths of some species extend between hemispheres. Wading birds are smaller than waterfowl but are the long-distance flight record holders. Satellite telemetry has revealed that bar-tailed godwits (*Limosa lapponica*) are capable of making nonstop flights of over 10,000 km from breeding territory in northern Alaska to New Zealand (Gill et al., 2005; Battley et al., 2012). Shorebirds mostly feed on mudflats or in shallow pools on marshes (Figure 7), but saltmarshes provide high-tide roosts. The availability of secure undisturbed roosting sites is important to minimize energy loss in birds preparing for long-distance flights. An essential attribute of roosting sites is that they provide uninterrupted sightlines so birds can detect potential danger; waders will not roost in tall vegetation. Tall vegetation, such as *Spartina alterniflora* or *Phragmites australis*, is habitat for secretive and rarely observed rail species (see *Shorebirds*).

Saltmarshes, Figure 7 Black-winged stilt *Himantopus himantopus* foraging amongst *Sarcocornia quinqueflora* at Sydney Olympic Park, Homebush Bay, Australia. Black-winged stilts occur in all continents except Antarctica. In Australia, most birds are resident but are mainly migrants in the tropical north.

Waterfowl and shorebirds are potential vectors for the transport, either internally or externally, of seeds (Proctor, 1968) and invertebrates (Frisch et al., 2007).

Many species of passerine birds are found on saltmarshes, and a particular feature of North America is the number of sparrow taxa in saltmarshes which are of conservation concern (Greenberg et al., 2006). In Britain, a large proportion of the world population of twite *Carduelis flavirostris* overwinter on saltmarshes, where they feed on seeds of a number of species (Norris, 2000). In Australia, one of the world's rarest birds, the orange-bellied parrot *Neophema chrysogaster*, overwinters on saltmarshes in the southeast mainland, feeding on the seeds of samphires.

Prior to forest clearance and the development of agriculture, saltmarshes were one of the few open habitats in temperate regions and may have been important refuges for large herbivorous mammals (Levin et al., 2002). Today on many marshes, the original large grazers have been replaced by livestock, but various species of deer utilize marshes, and, in Australia, saltmarshes are locally grazed by large kangaroos and wallabies.

Small mammals, such as rabbits and hares, may be obvious on some marshes. Other species such as voles can be abundant but remain well hidden. North American marshes are notable as habitat for a number of small rodent taxa with limited geographic distributions which are now regarded as threatened (Greenberg et al., 2006).

One group of mammals frequently ignored in assessments of saltmarshes is bats. Spencer et al. (2009) have documented the utilization of saltmarshes by Microchiroptera feeding on the abundance of flying insects. As many bat species are of conservation concern, this finding suggests that investigation of bats on saltmarshes should be undertaken more widely.

Invasive species

Invasion of ecosystems by introduced species is one of the factors seen as a major threat to biodiversity in most of the world's environments (see *Introduced Species*, *Invasive Species*). In the marine environment, estuaries are recognized as being particularly exposed to invasion (Levin and Crooks, 2011), and shipping is the major vector for introductions, either through ballast (historically from stone ballast, more recently in water ballast) or as fouling organisms on ships' hulls. An issue with detecting introductions is lack of reliable baseline data for many regions and for many taxonomic groups. For example, it will be difficult to determine whether many microalgae or microorganisms in estuaries are indigenous or introductions. Increasing population numbers of a species might be suggestive of recent arrival, but recent arrival does not prove introduction. Natural processes of dispersal and establishment did not end with the advent of humans. An increasing population could have been triggered in indigenous species by changes in the environmental conditions. Modern molecular genetic techniques may assist in both detecting and determining the status of introductions, but their application requires adequate and appropriate sampling. There has been controversy over whether the gastropod *Littorina littorea* on the eastern American seaboard was an introduction from Europe or whether unique American haplotypes indicate native status. The apparently unique American haplotypes are probably a consequence of under sampling, and a number of lines of evidence make a very strong case for introduction (Chapman et al., 2008).

In the case of vascular plants, the very wide, unquestionably native, distribution of many species in saltmarsh and mangroves might suggest that species had reached the limits of distribution through natural means, with little scope for further introduction. The deliberate introduction and subsequent establishment of mangroves in Hawaii and California disprove the hypothesis (Sauer, 1988). In the case of saltmarshes, large numbers of introduced plants have been recorded. Many of these are annual species, recorded from upper saltmarshes, particularly in Mediterranean climate regions. Whether these species have major biological impacts is uncertain. There is, however, a smaller number of perennial species which undoubtedly have major impacts. The two most important examples are the genus *Spartina* and *Phragmites australis*.

Saltmarshes, Figure 8 *Spartina maritima* at Knysna lagoon, South Africa, with a Grey Heron *Ardea cinerea.*

Spartina is a genus of halophytic grasses which is most diverse in the Americas. One species, *S. maritima*, is apparently native in the Old World, occurring in both Europe and Africa (Figure 8). *Spartina* species are now found in temperate and subtropical saltmarshes worldwide as a result of numerous documented deliberate introductions and natural spread from points of introduction (Strong and Ayres, 2009). The introduced *Spartina* spp. can occur lower on the shore than other species, so one of the reasons for introduction was to stabilize mudflats. A consequence has been a loss of intertidal mudflats, an important habitat in their own right.

One of the features of *Spartina* is the potential for interspecific hybridization. This is a chance event, with only a low probability of occurrence. However, successful hybridization between the American *Spartina alterniflora* and the indigenous *S. maritima* occurred in Southampton Water on the English Channel coast in the nineteenth century, initially producing the sterile *S.* x *townsendii* which subsequently gave rise to the fertile *S. anglica* (Strong and Ayres, 2009). *S. anglica* is now the low marsh dominant on much of the North European coast and also occurs widely elsewhere as a result of deliberate introductions (Ranwell, 1967). In the 1970s, the US Army Corps of Engineers introduced the eastern American *S. alterniflora* into California. Hybridization with the West Coast native *S. foliosa* occurred; the hybrid is rapidly spreading and through direct competition and continued pollen flow threatens the survival of *S. foliosa* (Strong and Ayres, 2009).

The common reed *Phragmites australis* has one of the widest natural distributions of any vascular plant, occurring in all continents except Antarctica. A European strain of *P. australis* was introduced into North America in the late nineteenth century and during the twentieth century spread widely, including into saltmarshes (Meyerson et al., 2009). Although native *P. australis* had occurred in upper saltmarshes in America, the European form is more salt tolerant (Vasquez et al., 2005), and this has been a factor in its spread. It now forms extensive near monocultures at many localities. A range of adverse consequences of this spread has been identified; however, it also offers potential benefits particularly through accumulation of peat, raising the ground surface and conferring protection against sea-level rise (Meyerson et al., 2009). Hybridization between native and introduced forms of *Phragmites* has been reported (Meyerson et al., 2010), further complicating issues.

Although there are strong similarities between the floras of inland and coastal saline habitats, particularly at generic level, there are differences. The genus *Tamarix* is native to Asia and Africa, in arid and inland saline habitats. A number of species of *Tamarix* were introduced into the United States in the nineteenth century (Di Tomaso, 1998) where they were used as windbreaks, ornamentals, and for erosion control. Subsequently, they have come to be regarded as amongst America's worst weeds. More recently, they have been reported invading coastal saltmarshes in California (Whitcraft et al., 2007). *Tamarix* spp. are shrubs to low trees, but they lack modified aerial root systems, so that even though they may convert saltmarsh to woodland, the new community formed would not be regarded as mangrove.

In regions where saltmarsh and mangroves coexist, expansion of mangroves into saltmarsh is occurring in several parts of the world (Saintilan et al., 2009), but most extensively in Australasia. This gives rise to questions for natural resource managers. Both saltmarsh and mangroves are recognized as being of high conservation value. Should action be taken to limit or prevent mangrove invasion, or should the loss of saltmarsh be allowed to continue?

Humans and saltmarshes

There is a long history of human utilization of saltmarsh resources (Adam, 1990; Gedan et al., 2009). Hunter gathering was (and, locally, is) widely practiced by indigenous peoples on saltmarshes around the world.

In northern Europe, many early agricultural settlements from the late prehistoric onwards were close to, or even on, saltmarshes. In the Netherlands, marshes supported grazing by livestock, were utilized for haymaking and for the collection of a variety of plant resources and even for cropping (Bakker, 1989; Meier, 2004; Knottnerus, 2005; Gedan et al., 2009). Similar practices were introduced during the colonial era into the Americas and Australia.

The agricultural use of saltmarshes has declined in many parts of the world, but it is still locally important (Figure 9). Harvesting plants such as *Salicornia* for human consumption has undergone a revival with the current interest in using wild-collected produce.

Grazing has considerable impact on vegetation, both because of direct effects on particular species and through trampling and compaction of the soil. The effect of grazing will depend upon the nature of the grazing animal – sheep, cattle, and horses have different

Saltmarshes, Figure 9 Sheep grazing on a saltmarsh in the Leven estuary, northwest England. Saltmarshes in this region have a long history of intensive grazing at high stocking rates (Gray, 1972). Saltmarsh lamb currently attracts a premium price at gourmet butchers' shops.

impacts – the timing of grazing, and the stocking rate. High stocking rates can result in a very short turf with low vascular plant diversity, but less intense grazing can result in taller vegetation and greater diversity. In Britain, the most diverse plant communities (Adam, 1981) are found at the upper levels of sites with moderate grazing pressure.

In northern Europe, species such as *Atriplex portulacoides* and *Limonium* spp. are grazing sensitive, and on heavily grazed marshes are either absent or restricted to microsites inaccessible to livestock, such as creek sides. Grazing results in permanent changes to vegetation. If livestock is removed, the vegetation does not revert to the community that would have been expected on marshes which had never been grazed. In general, when livestock grazing ceases the trend is for the development of tall, rank very species poor grassland communities (Bakker, 1989; Adam, 1990; Lambert, 2000).

The effects of livestock grazing are not only reflected in structure and composition of vegetation, but also by the fauna, in utilization by birds, with herbivorous waterfowl favoring shorter grass swards over taller vegetation, and the composition of invertebrate communities. Spider and beetle communities have been shown to be affected by grazing; abundance of some spaces is increased by grazing, but under intense grazing, spider species richness declines (Petillon et al., 2007).

Saltmarshes on the east coast of the United States were heavily exploited as grazing land and haymaking from early colonial times until relatively recently (Gedan et al., 2009). However, the legacy of centuries of human modification is not as obvious as it is in Europe. Why this is so is not clear, but perhaps the structural and floristic simplicity of *Spartina* marshes limited the changes resulting both from agricultural use and its subsequent decline. Ewanchuk and Bertness (2004) suggest that forb panne habitat in upper marshes in New England was reduced in extent by the effects of ditching which for three centuries was used to facilitate livestock grazing, haymaking, and mosquito control.

With the decline in direct exploitation of saltmarshes, they are increasingly valued for their ecosystems services, conservation, and for aesthetic experiences.

Threats

Despite increasing recognition of the values of saltmarsh, the ecosystem is under increasing threat worldwide. The range of threats facing saltmarshes has been identified in a number of publications (Kennish, 2001; Adam, 2002; Adam et al., 2008; Silliman et al., 2009b) and need not be recounted here (see *Anthropogenic Impacts*).

The majority of the world's human population is now urban, and many of the world's largest cities are on or close to estuaries. Even if environmental regulations are in place, there is a continuing increase in environmental degradation caused by accidental spills of chemicals in the catchment, the accumulation of pollutants from industrial discharges and domestic sources, eutrophication from general catchment runoff, stormwater inputs, and greater pressures for recreational use. As the world population grows and becomes more affluent, there is increasing need for greater development of ports and construction of airports, oil and gas terminals, and other industrial facilities. Ports, of necessity, must have waterfront access, while for many other forms of development, availability of level land created by marsh infill close to cities is seen as a major economic benefit. Construction of major airports on fill is viewed as a means of securing locations where noise impacts on residential suburbs can be minimized. The importance of saltmarsh habitat loss from major infrastructure development is recognized, but the economic and social benefits (at least in the short-term) of development prevail over environmental concerns.

Development pressures are localized in their impacts, even if cumulatively the effects are large, but the growing human population, increasing living standards, and development will result in continuing release of greenhouse gases and consequent climate disruption. One of the consequences of global warming will be an increase in the volume of the oceans due to thermal expansion and, in the longer term, the melting of ice caps and glaciers. A potential consequence of sea-level rise will be "drowning" of saltmarsh and "coastal squeeze," where natural topography or artificial constraints prevent compensation for loss of the seaward parts of marshes by retreat landward. On a site-by-site basis, what is important will be relative change in sea level; the eustatic rise may be compensated for by sedimentation or isostatic or tectonic change in land level. In some circumstances, the increase in atmospheric CO_2 may lead to increased plant production and greater incorporation of organic material into saltmarsh soils (Langley et al., 2009) (see *Sea-Level Change and Coastal Wetlands*).

The role of saltmarsh in sequestering carbon in soil is of global significance (Chmura et al., 2003, Chmura, 2009). The importance of "blue carbon" in ameliorating the effects of greenhouse gas emissions is likely to be of increasing importance in the future.

Increased temperature may affect plant distribution and evapotranspiration and hence soil salinity. Changes in other aspects of climate such as rainfall patterns or the incidence of major storms will also have impacts on saltmarshes, some local and some more widespread. The full range of climate change impacts is difficult to predict except in general terms (Ross and Adam, 2013), but it is clear that there will be changes in both the physical and biotic components (Semeniuk, 2013). Increase in CO_2 will affect the relative competitiveness of plants, with C_3 species likely to have an advantage over C_4 (Mayor and Hicks, 2009) species. Hence, changes in the floristic composition of saltmarshes in temperate and subtropical marshes can be anticipated, with concomitant changes in fauna. *Spartina* spp. are currently the most widespread C_4 species on saltmarshes (see *Climate Change*).

Values

As part of the evaluation of biodiversity, there has been recognition of the importance of biodiversity in the provision of ecosystem services, processes, and functions. The importance of saltmarshes for the maintenance of fisheries has been recognized for more than half a century; other services, although long recognized anecdotally by local residents, had only recently been scientifically assessed. Barbier et al. (2011) have reviewed the services provided by saltmarsh and attempted an economic evaluation. Economic data are available from a limited number of saltmarsh types and sites. While the data demonstrate high values in the instances studied, extrapolation to saltmarshes as a whole requires caution.

Recent major storms and recognition that the incidents and intensity of storms are likely to increase in some localities as a result of climate change that has promoted interest in the role of saltmarshes in absorbing wave energy and protecting the hinterland (Costanza et al., 2008; Koch et al., 2009). Shepard et al. (2011) have reviewed the role of saltmarshes in protecting the coastline against storms. Vegetation cover plays a major role in attenuation of wave impact, while belowground biomass is important in limiting erosion. Yang et al. (2012) demonstrated that *Spartina alterniflora*, an introduced species in China, was more effective in attenuating wave energy because of its height than the shorter native species. Whether, given the potentially adverse impacts of *S. alterniflora* on Chinese saltmarshes (An et al., 2007), continued planting of *S. alterniflora* is justified is a policy question for managers. Tall *S. alterniflora* is the dominant species in the Gulf and Atlantic coasts of the United States and may perform a greater role in wave attenuation than the short vegetation of most marshes elsewhere. The magnitude of attenuation is influenced by storm intensity and water depth, but it is clear saltmarsh can protect the coastline.

The old concept of "wetlands as wasteland" is dead and buried. The importance of ecosystem services provided by wetlands is now firmly established. More intangible, but no less important, is the aesthetic value of saltmarsh, long recognized by landscape painters and increasingly so by a much wider public.

Conservation and management

Saltmarshes have long been valued for conservation. Some of the earliest formal major conservation reserves were saltmarshes (Davy, 2009), which have been perceived as valued sites for research and community education and as places of beauty. The large numbers of birds associated with saltmarshes have also been a major factor behind the designation of many saltmarsh sites as conservation reserves and the listing of saltmarsh sites as wetlands of international importance under the Ramsar Convention. Saltmarshes are afforded special protection in a number of jurisdictions (e.g., in the Australian state of New South Wales coastal saltmarsh is an Endangered Ecological Community, while the European Union Habitats Directive recognizes a range of saltmarshes as requiring designation as Special Areas of Conservation).

Conservation of saltmarshes will not be achieved simply by designating sites as reserves. Given the range of threats saltmarshes face, there will need to be active management. Management regimes for saltmarshes need to be integrated into wider coastal zone management plans (Shepard et al., 2011), and given the importance of saltmarshes for migratory birds (and in the case of waterfowl the adverse impacts they may have on saltmarshes), there needs to be an international component to management plans. Species of fish, for which saltmarsh is nursery habitat, may distribute widely as adults, so national and international management of marine resources will also need to consider saltmarsh. Management of sediment supply and hydrology requires saltmarsh management to be integrated into catchment management.

Over the past half century, there has been increasing attention paid to rehabilitation, restoration, and recreation of saltmarshes (Zedler, 2001; Roman and Burdick, 2012). Substantial sums have been expended on this work, which has frequently been mandated by planning and environmental laws. Considerable progress has been made in the field, but there is still debate as to how success of programs should be measured and whether or not it is achieved (Ambrose, 2000; Mossman et al. 2012a; Mossman et al. 2012b).

Summary

Saltmarsh is the natural fringing vegetation of many estuaries, most particularly in temperate and higher latitudes, but saltmarsh also occurs in conjunction with mangroves in tropical and subtropical regions. The development of saltmarsh involves dynamic interactions and feedbacks between the biota and the physical environment. Saltmarshes provide habitat for a rich biota. The ecosystem services provided by saltmarshes include sustaining fisheries, provision of habitat migratory birds, provision of shoreline protection, and sequestration of carbon. Anthropogenic pressures on estuarine resources and the effects of increased carbon dioxide and climate disruption will pose threats to the long-term sustainability of many saltmarshes unless coordinated international action can be undertaken.

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Cross-references

Anthropogenic Impacts
Climate Change
Coastal Wetlands
Estuarine Habitat Restoration
Introduced Species
Invasive Species
Mangroves
Salt Marsh Accretion
Sea-Level Change and Coastal Wetlands
Shorebirds
Species Zonation

SAND MINING/BEACH SAND MINING

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Definition

Extraction of sand from, typically, a natural deposit, for use at a different location. It is most often performed for land reclamation, beach nourishment, or procurement of material for construction (particularly for concrete) or industrial uses (such as sandpaper or foundry supply). The sand source is often a terrestrial dune or hill, a riverbed, a beach, or an offshore deposit. Spoil resulting from dredging operations conducted for other purposes can be used beneficially, representing a form of sand mining.

Introduction

Sand is heavily utilized in the construction industry as a major ingredient in concrete (often 25 % by volume; Kosmatka and Wilson, 2011). Being widely available, inexpensive, durable, and even pourable, it also has many other industrial uses, particularly as part of an abrasive system such as sandpaper. It is widely used in foundries and for making glass and ceramics.

In some cases the harvested sand is processed for extraction of selected minerals, including titanium and zirconium, and, on selected Namibian beaches, diamonds. Many of the world's beaches are comprised primarily of quartz and feldspar sands, but it is not uncommon to find other minerals of value such as zircon (zirconium oxide) mixed in with these grains. The sands can be mined to remove the heavy minerals of interest in such a case.

Sand is still mined from beaches, but this practice is prohibited in many locations because it results in what is essentially anthropogenic beach erosion and is therefore undesirable. Environmental issues such as the effects of enhanced turbidity or exposure of contaminants that arise from mining activities should also be considered before undertaking sand mining (e.g., Meador and Layher, 1998).

Since sand is found, and used, worldwide, sand mining also takes place on a global scale. Different types of sands are utilized for different purposes (concrete, foundry sands, etc.). Most of the marine sand mining occurring on the US Atlantic and Gulf coasts is for beach nourishment purposes. It has been estimated that the volume excavated for this purpose in the twentieth century is on the order of 650 million cubic meters (Finkl and Hobbs, 2009). This is larger than the volume of Lake Erie.

Techniques

A wide variety of techniques can be employed for sand mining. Marine sand mining is usually accomplished with the same types of equipment used for dredging projects, including clamshell buckets, backhoes, dustpan, and cutterhead suction dredges (see *Dredging*). The hydraulic approaches require dewatering of the sand-water slurry but can achieve higher rates and facilitate transport by allowing pumping through long pipelines to desired discharge points. Terrestrial deposits can likewise be excavated by a wide variety of techniques, but most are classified as surface mining, digging from the top of the deposit downward, continually excavating the exposed surface of the deposit.

Summary

Sand mining is a major, global industry and will remain important for many years to come. It has been utilized historically to acquire building and industrial supplies, but new applications arise over time, such as excavation of tar sands for their energy value and the mining of frac sands for use in hydraulic fracking operations to extract subterranean hydrocarbons. Likewise, new environmental issues to be mitigated are likely to continue to arise. Terrestrial sand mining in particular tends to lead to protests from citizens concerned with land use and environmental issues, because it often results in highly visible, permanent changes to the landscape. The western Great Lakes region in the United States (Wisconsin, Minnesota), for example, is the source of a large amount of sand being used for fracking operations and is dealing with public protests over the mining of these materials.

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Cross-references

Anthropogenic Impacts
Dredge and Fill
Dredging
Mass Physical Sediment Properties

SAND RIDGE

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Synonyms

Linear shoal; Sand shoal; Shoreface ridge; Tidal current ridge

Definition

Sand ridges are elongate coastal or shelf sand bodies that form bathymetric highs on the seafloor (Duane et al., 1972).

Description

Sand ridges are asymmetrical features composed of unconsolidated fine to coarse sand and gravel. They are typically 5-120 km long and 0.5-8 km wide, with heights of more than 20 % of the water depth and slopes of less than 1°. They occur on continental shelves in a wide range of water depths, where sufficient sand exists and currents are strong enough to transport sand-sized sediment. They also usually occur in groups that are shore parallel, shore oblique, or shore normal, with spacing between members of ~250 times the water depth, along topographically irregular transgressive margins (Snedden and Dalrymple, 1999). Based on the relative progression of ridge reworking and migration (Snedden and Dalrymple, 1999), sand ridges belong to one of three classes. Class I ridges are young ridges that maintain their original nucleus and are at or near their point of origination. Class II ridges are partially evolved ridges that have maintained some of their original nucleus and have migrated a distance less than their original width. Class III ridges are fully evolved forms that have maintained none of their original nucleus. They have migrated a distance greater than or equal to their original width.

Several hypotheses and models have been developed that address the origin of sand ridges; these are largely based on their distribution and orientation. They include (1) the drowning of static, pre-transgressive features such as barrier islands, beach ridges, and interfluves (e.g., Veatch and Smith, 1939); (2) the formation of post-transgressive features in equilibrium with the hydraulic regime (e.g., Uchupi, 1968) or formed during storms (e.g., Duane et al., 1972; Swift and Field, 1981); and (3) the reworking of modern features associated with an actively transgressing shoreline (e.g., Swift and Field, 1981; Robinson and McBride, 2008).

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Cross-references

Bar
Barrier Island
Barrier Spits
Coastal Barriers
Coastal Landforms
Sedimentary Structures

SANDBANKS

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Synonyms

Bank; Hummock; Ridge; Sands; Shoal

Definition

Sandbanks are bedforms consisting of cohesive or non-cohesive sediments in estuaries and continental shelf areas that can be exposed at low tide. They are generally formed by sediment transport by tidal currents and waves as well as changes of tidal prism due to sedimentation in estuaries.

Description

Sandbanks have been the focus of an array of studies (e.g., Dyer and Huntley, 1999; Christopherson, 2002; Chaumillon et al., 2008; Reeve et al., 2008; Brown and Davies, 2009; Neill and Scourse, 2009). They mainly consist of sand particles with sizes varying from very coarse (2 mm–1 m diameter) to very fine (0.10–0.05 mm) but also can include silt (0.05–0.002 mm), clay (<0.002 mm), and biological matter. Typical locations of sandbanks are ebb and flood deltas of tidal inlets within and in the vicinity of estuary mouths and near coastal headlands.

Sandbanks generally form from tidal flows through straits, past headlands and islands, where the currents are rapid. At the ends of these features, the currents decrease, and sediment deposition occurs. Sandbanks develop from sand convergence driven by both tidal currents and waves. In estuaries, sediment infilling causes the decrease in the tidal prism on a century and millenia time scale, leading to sandbank formation.

The location of sandbank formation has been used for their classification. For example, sandbanks may be classified as open shelf linear sand ridges (Type 1), wide estuary sand ridges (Type 2A), ebb and flood deltas (Type 2Bi), shore attached ridges (Type 2Bii), banner banks (Type 3A), and en-echelon ridges (Type 3B).

Sandbanks play a key role in natural coastal defenses from waves (e.g., at the mouth of the Dyfi estuary in North Wales), navigation, and as resources for sediment aggregates. Additionally, sandbanks are prime locations for offshore wind farm deployments in coastal regions. They are also used as nesting grounds by sea lions and as feeding and resting grounds by coastal birds such as seagulls and oyster catchers.

There are many excellent examples of sandbanks. Two of these in the UK include the Great Yarmouth sandbanks and the Isle of Portland sandbanks. The Great Yarmouth sandbanks may have been formed when the sea transgressed from the south through the Dover Strait at the end of the Holocene period. Some deep channels between the sandbanks were also formed due to strong tidal currents. Two of the major sandbanks of the Isle of Portland (West Shoal and Adamant Shoal) may have been formed when the Isle of Portland was isolated from the mainland by a navigable strait. After sedimentation caused the closure of the strait, these two sandbanks were no longer actively maintained and subsequently migrated towards the two largest present-day headland sandbanks of the region (Portland Bank and Shambles Bank).

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Cross-references

Deltas
Mass Physical Sediment Properties
Sediment Transport

SANDFLAT

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Synonyms

Mudflat; Tidal flat

Definition

A sandflat is an extension of unconsolidated sediment located preferentially in the lower intertidal zone. It is an unstable area characterized by the constant resuspension of sediment by tidal flood and ebb currents. Tidal forces predominate over other hydrodynamic processes, forming well-sorted sand deposits.

Hydrodynamics

The amount of mud found in the sediment of a sandflat is determined by the bed slope, the amount of fine sediments derived from land, and the strength of the current. The sandflat-mudflat threshold is determined by the aforementioned factors. Finer muddy sediments accumulate during neap tides, while coarser sandy sediments accumulate during spring tides. Hydrodynamic conditions of sandflats preclude development of submerged aquatic vegetation; however, there are important resident communities of burrowing invertebrates (worms, bivalves, crustaceans) that are fundamentally important for local trophic networks and the recycling of nutrients. Sandflats can be found adjacent to other types of habitats, such as saltmarshes located in the upper intertidal zone (Perillo, 1996; Schwartz, 2005; Perillo et al., 2009).

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Cross-references

Sandbanks
Sediment Grain Size
Sediment Transport
Tidal Flat
Tidal Flat Salinity Gradient

SEABIRDS

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Synonyms

Estuarine birds; Marine birds; Pelagic birds

Definitions

Seabirds. Birds that live primarily on the oceans of the world and breed on oceanic islands, although it includes some members of seabird families that breed inland.

Colonial. Several to hundreds of birds nesting in close proximity.

Introduction

Seabirds are at home on land, in the air, and in water, making them unique among vertebrates. They often switch between these three environments on a daily basis, which requires unique physiology and morphological adaptations. They have adapted to all ecosystems on earth, from the Arctic to Antarctica. Seabirds can be defined as those that live, forage, and breed in marine environments, including bays, estuaries, wetlands, coastal islands, oceanic islands, and the open ocean. They mainly forage at sea, either nearshore or offshore, and can spend days, weeks, and even years at sea without coming to land. Once albatrosses leave their natal colony, they may wander the oceans for 5–10 years before coming to land to breed.

While the ocean may seem uniform, seas vary with seasonal cycles, El Nino events, and stochastically. The distribution and foraging behavior of seabirds are interconnected with oceanographic influences. For example, 50 species of seabirds are regular residents of the tropical Pacific Ocean, where specific currents, distinct water masses, El Nino events, and the presence of marine mammals all affect seabird distribution (Balance et al., 2006). Many seabirds forage with predatory fish or marine mammals because they force prey fish to the surface, making them available to foraging seabirds.

Relevant life history

Seabirds are long-lived (20–60 years), delay breeding (some albatrosses do not breed until they are 10 years old), lay small clutch sizes (most lay only one egg), have extended parental care (weeks to 6 months or more), and range in size from a 0.24 kg Snow Petrel (*Pagodroma nivea*) to a 16 kg male King Penguin (*Aptenodytes patagonicus*). Thus, seabirds have low clutch sizes and long-parental care, producing enough young to replace

Seabirds, Figure 1 Great and Snowy Egrets nesting in *Iva low bushes in Barnegat Bay, New Jersey.*

themselves during their lifetimes. Even within seabirds, life history strategies differ. Gulls and terns breed when they are 3–5 years old, lay two to three eggs, have incubation periods of a few weeks, produce a young or two a year, and live only 20 years, whereas albatrosses do not breed until they are 8–10 years old, lay one egg, do not raise a young every year, and may live up to 60 years. David Lack (1968) first proposed that the life history strategy of seabirds evolved because of energy limitations that derive from the difficulty of feeding over the open ocean where prey is unpredictable and irregular (Schreiber and Burger, 2001a) (Figure 1).

Seabirds often nest in colonies of a few to several thousand pairs in places that are free of predators, such as offshore islands. Some seabirds have altricial young (must be feed from their parent's bill, naked when born, e.g., Pelicans), while others have precocial chicks (covered with down, able to walk immediately upon hatching, e.g., gulls). Both types require long periods of food provisioning and predator protection while they are in the nest and for weeks or months thereafter (Schreiber and Burger, 2001b, c).

Habitat selection

Breeding habitat

Seabirds nest on land and forage over water in estuaries, bays, along coasts, or in offshore waters hundreds of kilometers from coasts (Table 1). Nesting seabirds have three habitat selection decisions: (1) whether to nest in a colony or solitarily, (2) what general breeding site to choose, and (3) what specific nest site to select. Most seabirds nest in groups called colonies, and numbers range from dozens in terns (e.g., Common Tern, *Sterna hirundo*) to thousands of pairs (Sooty Tern, *Onychoprion fuscatus*). The advantage of nesting solitarily is that cryptic birds can nest in vegetation and leave the nest quietly, thereby avoiding predators, while the disadvantage is that there are no colony mates to provide early warning, help defend nests, or provide information on food sources. The advantage of nesting in colonies is that colony mates provide early warning, colony defense, and information about food sources. In addition, any bird in the colony has less of a chance that it will be targeted by a predator because the predator has many birds to choose from (predator swamping). If a solitary nesting seabird is located by a predator, it may be killed (Burger, 1982; Coulson, 2001).

The second decision, where to locate a colony, usually results in nesting in places that are inaccessible to mammalian predators, such as oceanic or offshore islands, cliffs, or trees. Once selected, colony sites usually remain stable unless they are destroyed by predators or people or become unusable because of habitat loss, high tides and flooding, disease, or other physical disruption. The third choice is where to defend a breeding territory and nest. Usually males select a territory and court a female. Both members of the pair then participate in territory defense, incubation, and chick rearing.

Since birds incubate their eggs, they are place based during the incubation period and must select a safe place free from predators that will also provide protection from inclement weather and stressful temperatures. Remote sensing and geographical information systems (GIS) allow seabird biologists to assess habitat availability and quality, monitor populations and colony use, and develop conservation plans (Gottschalk et al., 2005).

Foraging

Seabirds mainly forage by swimming underwater to pursue prey, sitting on the water and picking up prey, dipping prey from water while flying above, and plunge diving for prey from well above the water (Table 1). In addition, some gulls frequent garbage dumps, pick up food from lawns or gardens, and scavenge dead fish or other prey along the shore. Species, such as gulls, that have very diverse foraging methods generally nest along coasts, have larger clutch sizes, raise more young, and have shorter lifespans than pelagic species that forage over the open ocean. While it might seem that fish and other prey are plentiful in oceans, prey is not evenly distributed and may be unavailable because birds cannot

Seabirds, Table 1 Orders of marine birds (After Burger and Schrieber, 2001a; Burger and Schrieber, 2001b; Brooke, 2001; Shealer, 2001, Unpubl. data). The number of species is shown in parenthesis after the family name (authors disagree on exact numbers)

Order	Family	Types of birds	Foraging	Nesting
Sphenisciformes	Spheniscidae (17)	Penguins	Underwater by pursuit diving	Ground or burrows; offshore or coastal islands, some on Antarctic islands
Procellariiformes	Diomedeidae (21)	Albatrosses	Surface dip or surface seize	Ground, oceanic islands
	Procellariidae (79)	Gadfly petrels, shearwaters, and fulmars	Hover dip or surface dip and pattering	Islands, burrows
	Pelecanidae (4)	Diving petrels	Diving	Burrows
	Hydrobatidae (21)	Storm petrels	Dipping, pattering, and surface dip	Islands or high cliffs; crevices, burrows
Pelecaniformes	Phaethontidae (3)	Tropic birds	Surface seizing	Crevices, rock burrows
	Pelecanidae (7)	Pelicans	Surface seizing Plunge diving	Ground, trees
	Frigatidae (5)	Frigate birds	Piracy, dipping	Trees
	Sulidae (10)	Gannets and boobies	Plunge diving	Ground, trees
	Subfamily (40) Phalacrocoracinae	Cormorants, anhingas, and darters	Pursuit diving underwater	Ground, trees pilings
Charadriiformes	Stercorariidae (7)	Skuas and jaegers	Dipping, piracy	Ground
	Subfamily Larinae (50)	Gulls	Dipping, surface seizing, and piracy	Ground, floating nests in marshes, and trees/shrubs
	Subfamily Sterninae (45)	Terns	Plunge diving, dipping, and piracy	Ground, shrubs
	Rhynchopidae (3)	Skimmers	Skimming	Ground
	Alcidae (23)	Auks	Pursuit diving underwater	Crevices, burrows, and ground

Note: Rhynchopidae are not usually considered seabirds because they may forage in estuaries, but never use the open ocean

swim that deep, plunge dive that far below the surface, or dip down far enough to catch fish.

Predators, competitors, and invasive species

In addition to food availability (both types and quantity), seabird survival and reproductive success are affected by predators and competitors. There are no mammalian predators on offshore or oceanic islands where most seabirds nest, and there are few avian predators so far from land. Other seabirds nest on cliffs or rocky ledges where mammalian predators have no access, and spaces are too small for avian predators to land. Seabirds that nest along coasts either nest on cliffs, islands within bays, or in trees where predators have little access. Their main competitors are other seabirds of the same or a different species that nest within these colonies. In some cases, space itself is limited, restricting the number of pairs that can breed. Stiff competition among Northern Gannets (*Morus bassanus*) for nest sites on rock outcroppings results in adults remaining several weeks or months after the young fledge, just to protect their territory (Mowbray, 2002).

Invasive species provide a unique threat because they did not evolve with seabirds, and they have fewer adaptations for coping with them. Invasive mammalian predators pose the greatest threat to seabirds worldwide (Nettleship et al., 1994). Seabirds nesting on oceanic islands evolved without predators and usually nest on the ground, where they are vulnerable to invasive mammals such as feral cats (*Felis catus*), rats (*Rattus* sp.), and mice (*Mus* spp.) that prey on eggs, young, or even adults. On many islands, such as the United Kingdom Overseas Territories, cats and rats have caused near extinction of several seabirds, resulting in critically endangered species (Hilton and Cuthbert, 2010). Examining 94 papers that demonstrated effects of rats on nesting seabirds, Jones et al. (2007) showed that 74 seabird species in ten families were affected. Storm Petrels and small, burrow-nesting seabirds were most affected, and gulls and terns were least affected. Removing rats and cats from islands with nesting seabirds should be a high conservation priority (Jones et al., 2007).

Migration and overwintering

Seabirds have some of the longest migration routes of any birds; but since they mainly migrate over oceans, their patterns were a mystery until very recently. With advances in radiotelemetry (shore distance), satellite transmitters (on large birds), and geolocators (for smaller birds), data on specific migration routes can be combined to form migration patterns for species. Most seabirds spend the winter at sea, never or seldom coming to land.

Threats to seabirds

Because seabirds inhabit so many habitats, in all ecosystems, they are exposed to many threats, including habitat loss, development, human activities and disturbance, disease, toxic chemicals, plastics, invasive species, and fisheries. Habitat loss is the primary factor affecting seabirds that nest along coasts because these regions are highly developed, putting pressure on the cliffs, islands, and bare ground where seabirds nest. In the long term, climate change may pose a great risk by affecting fisheries bycatch, spatial ecology, food sources, and prey availability (Gremillet and Boulinier, 2009; Barbraud et al., 2012).

Seabirds are exposed to a range of human activities that disrupt their breeding, foraging, migrating, and overwintering. Disruptions include direct disturbance, such as people entering breeding colonies, scaring incubating adults from nests, destroying eggs, bringing in predators (e.g., dogs), and disrupting foraging with boats, Jet Skis, or other water-related activities. Indirect effects include increases in native predators because of availability of food brought by humans (e.g., raccoons). Effects on nesting and foraging seabirds can occur as a result of scientific investigators or others that visit colonies or monitor seabird behavior. A meta-analysis of the effect of transmitters on avian behavior found significant effects – birds with devices had greater energy expenditures and some failed to nest (Barron et al., 2010). However, usually devices are used on only a few individuals, and information gained from these is greater than the cost to these individuals and any potential costs of alternative methods of learning about movements and migration patterns (Burger et al., 2012).

Fisheries are a special case of human activities; fisheries have a positive and a negative effect. Positive effects include provision of food from offal, around factor ships, or near processing plants. Negative effects include mortality from capture in fishing lines or equipment or being entangled and drowning (called bycatch, Moore et al., 2009) or because of competition with fisheries for fish. Prey depletion can be a problem during breeding when adults are limited to foraging distance around their nests. Seabirds and fisheries have interacted for centuries with little effect until the rapid enhancement of fishing capabilities and overexploitation of fish stocks that happened in the last two centuries. However, the advent of large factory ships and deployment of longlines that extend for kilometers increased overfishing of fish stocks and massive increases in captures in nets, especially albatrosses and petrels (Montevecchi, 2001).

Human activities are responsible for most of the toxic chemicals in the environment, and where toxic elements are present in soil or water (e.g., mercury in seawater), organisms have adapted to natural levels. Pollution in the ocean comes from dumping of wastes, dredging, runoff from towns and rivers, and point source pollution from factories, urbanization, suburbanization, and coastal businesses and activities. Toxics and plastics threaten seabirds because they spend so much time in water where they are exposed externally, by inhalation and by ingestion of food and water (Burger and Gochfeld, 2001). Both individuals and populations can be affected, either acutely or by chronic exposure. The chemicals that have been shown to affect seabirds include lead, mercury and other metals, oil, polychlorinated biphenyls (PCB), dichlorodiphenyltrichloroethane (DDT), and other organochlorine compounds (Burger and Gochfeld, 2001). Oil spills can cause large-scale mortality (Kingston, 2002). DDT is the classic case of toxic effects. Populations of several fish-eating birds (e.g., Brown Pelican, *Pelecanus occidentalis*; White Pelican, *P. erythrorhynchos*; and Gannets) collapsed in the 1960s because DDT interfered with calcium and eggshell development (Blue et al., 1974). When adults sat to incubate, their thin-shelled eggs broke. Plastics are synthetic organic polymers, and their increase over the last two decades resulted in large quantities of plastics entering the oceans. Seabirds are mainly affected by entanglement, exposure to PCBs from ingestion, and compromises to the digestive tract because of ingestion of large quantities (Derraik, 2002). Many species of seabirds are endangered (Nettleship et al., 1994; Schreiber and Burger, 2001a).

Summary

Seabirds, including penguins, albatrosses, petrels, boobies, pelicans, gulls, and terns, spend most of their life foraging at sea and breed on coastal or oceanic islands far removed from mammalian predators. They are long-lived and have delayed breeding, low clutch sizes, long incubation periods, and extended parental care. Most nest in colonies that are either monospecific or contain many different species. Nesting in colonies allows for early warning, groups defense, predator swamping that reduces the chances of any individual being eaten, and information transfer about food sources. The threats to seabirds include habitat loss and degradation, overfishing, invasive predators, toxics, plastics, and climate change. The biggest threat to nesting seabirds is nonnative mammalian predators, while fisheries provide the biggest threat to foraging seabirds (both mortality and competition for prey). Many species of seabirds are endangered.

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Cross-references

Shorebirds

SEAGRASS PRODUCTION MODELS

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Synonyms

Ecological models

Definition

Seagrass production models are conceptual or mathematical simplifications (or abstractions) of the physiological processes associated with plant metabolism that results in plant growth or loss.

Introduction

Seagrasses, submersed marine angiosperms, are important components of global shallow coastal and estuarine ecosystems (Green and Short, 2003). Seagrass communities provide habitat, protection, and nursery functions for economically valuable fishery species (Heck et al., 2008), serve as indicators of and modify local water quality conditions (Dennison et al., 1993; Moore, 2004), and link nutrient and carbon cycles between the water column and sediment (Fourqurean et al., 2012). Due to their importance within shallow coastal ecosystems, information concerning the growth, resilience, and stability of seagrass beds is necessary for effective coastal management.

Ecological models are conceptual or mathematical simplifications (or abstractions) of a real system (see *Ecological Modeling*). Production models are a type of ecological model that mathematically describes the physiological processes associated with plant metabolism that results in plant growth or loss (Best et al., 2001). These models are useful tools in the quantitative analysis of complex ecosystems, such as seagrass meadows, and aid in the development of hypotheses of feedback mechanisms that impact plant growth and predict how plants may respond to changes in water quality or management practices that cannot be quantified using field and laboratory data alone (Carr et al., 1997).

Primarily seagrass production models have been used for scientific research focusing on quantifying physiological responses to environmental conditions (Short, 1980; Wetzel and Neckles, 1986; Bach, 1993). More recently the scope of production models has shifted to investigate broader topics, such as altered trophic interactions (Zaldívar et al., 2009; Baeta et al., 2011), and has been coupled to hydrodynamic (Manca et al., 2012) and/or spatially explicit models (Coffaro et al., 1997; Giusti et al., 2010) to analyze the ecological roles and responses of seagrass populations across larger scales. With increasing availability of long-term monitoring data sets and the continued decline of seagrasses globally (Orth et al., 2006;

Waycott et al., 2009), the role of seagrass production models has shifted to more of a synthesis-, forecast-, and management-driven focus (JØrgensen, 1994).

Framework of seagrass productivity models

Mechanistic seagrass productivity models follow the flow of carbon, the product of primary production, through a modeled environment and track the flow as it is modified by the defined model parameters (Short, 1980). Most mechanistic seagrass production models follow a mass-balance approach to quantify the change in a seagrass population over time as a function of the rate of biomass production from gross photosynthesis and loss due to respiration, mortality, and herbivory (Madden and Kemp, 1996; Cerco and Moore, 2001). Mass-balance equations are set up for all major, or state, variables and are collectively known as governing equations. Other examples of common state variables found in seagrass production models include epiphytes, phytoplankton, macroalgae, zooplankton, and various fish species (da Silva and Asmus, 2001; Biber et al., 2004; Baeta et al., 2011).

Modeling environmental factors that influence seagrass production

Primary production is the result of gross photosynthesis, which is the rate at which organic carbon and oxygen are produced through the conversion of light energy into chemical energy (Marker and Westlake, 1980). Gross photosynthesis (*P*) is often modeled as a function at a -species-specific optimal photosynthetic rate (P_{max}) under optimal or known fixed environmental conditions (Vermaat and Hootsman, 1994; Madden and Kemp, 1996; Cerco and Moore, 2001). Parameters that modify the rate of processes in seagrass production models should have a measureable impact on seagrass ecosystems, contain ecologically relevant equation coefficients, and avoid empirical equations as much as possible (Best et al., 2001). While the abiotic factors that influence P_{max} vary, the availability of light, water temperature, availability of nutrients, and initial plant biomass are often included in most seagrass production equations (Carr et al., 1997).

Light

The primary factor impacting photosynthesis in coastal seagrass ecosystems is light availability (Ralph et al., 2007). However, not all light that reaches the surface of the water within a seagrass bed is available for photosynthesis (Dunton and Tomasko, 1994; Zimmerman et al., 1994). For most seagrass production models, the availability of light at the leaf surface depends on (1) the total amount of available light, (2) the amount of light reflected at the surface of the water, (3) water column light attenuation, (4) the amount of dissolved and suspended particles in the water column, and (5) light reduction by epiphytes on the leaf surface (Madden and Kemp, 1996).

Photosynthetic rates, described in the literature using species-specific photosynthesis-irradiance or PI curves, increase linearly with light up to a saturating level past which photosynthesis no longer increases (Lee et al., 2007). In several seagrass production models, the relationship between light and photosynthesis in seagrass models is often defined by the Michaelis-Menten function or the hyperbolic tangent function (Vermaat and Hootsman, 1994; Zimmerman et al., 1994; Madden and Kemp, 1996). Both functions assume that photo-inhibition does not occur in seagrass beds and reduce P_{max} either with a light half-saturation constant (Michaelis-Menten) or by a light-saturation threshold (hyperbolic tangent equation) to account for the impact of low light conditions on photosynthesis.

Temperature

Temperature impacts seagrass production by controlling the rate of chemical reactions within the plant (Lee et al., 2007). For most biological processes, there is an optimal temperature range at which the process is the most efficient (Thornton and Lessem, 1978). As temperatures depart from the optimum range, the rates of processes will change, impacting overall production (Lee et al., 2007). The relationship between temperature and physiological processes is species, location, and seasonally specific (Orth and Moore, 1986; Thom, 1990). Therefore no single algorithm works for all species, and the selection of the equation used to describe the impact of temperature on seagrass productivity should be based on a wide range of temperatures (Carr et al., 1997).

Nitrogen and phosphorus

Many seagrass production models also consider the impact of nutrients on gross photosynthesis as an important growth-limiting factor (Zimmerman et al., 1987; Bocci et al., 1997; Lee et al., 2007). Seagrasses are capable of obtaining nutrients from both the water column and the sediment, and uptake of nutrients by both roots and shoots should be included when possible (Lee et al., 2007). Traditionally the Michaelis-Menten equation has been used to model seagrass production response to nutrient limitations. The equation uses an estimate of the half-saturation constant that corresponds with the nutrient concentration where productivity is one-half P_{max} (Madden and Kemp, 1996; Cerco and Moore, 2001). Half-saturation constants will also vary between the roots/rhizomes and leaves, and all values are species specific.

Modeling environmental factors that influence seagrass biomass loss

Loss of seagrass biomass can be attributed to factors such as physical disturbance, microbial decay, herbivory, high rates of respiration, and mortality (Short, 1980; Madden and Kemp, 1996; Cerco and Moore, 2001). While the impacts of environmental factors such as temperature, dissolved oxygen, and nutrients on respiration have been investigated, the mechanistic relationships underlying these impacts are not well known (Marsh et al., 1986; Hemminga, 1998; Clavier et al., 2011). Therefore, many seagrass models use empirical relationships or set loss rates equal to a constant proportion of plant biomass to drive processes like respiration and mortality that result in biomass loss (Bocci et al., 1997; Carr et al., 1997; Cerco and Moore, 2001). Additional research into the physiological response driving the mechanisms that result in the loss of seagrass biomass is needed to accurately model these important processes.

Current knowledge gaps

Beyond the factors that drive rates of production and loss within seagrass beds, future production models need to consider the complete life history strategy of seagrasses. Exclusion of sexual reproduction in carbon-based models has been accepted due to the dominance of vegetative reproduction in seagrass beds and the relatively low carbon value of seeds (Harwell, 2000). However, recent research has shown that sexual reproduction plays a significant role in seagrass bed resilience and expansion (Kendrick et al., 1999; Yang et al., 2013) and recovery from large-scale declines (Plus et al., 2003; Jarvis and Moore, 2010); therefore, key components of the bed recovery and expansion dynamics may be missing from seagrass production models when sexual reproduction is excluded. In particular, information on seedling physiology is necessary to accurately parameterize the response of seedlings to environmental conditions.

Future production models should also broaden their focus beyond the impacts of light, temperature, and nutrients and further integrate sediment conditions through the development of sub-diagenetic models (Eldridge and Morse, 2000). Sediment characteristics including organic content, redox conditions, grain size, nutrient concentrations, and sulfide levels have all been shown to impact seagrass growth and survival (Goodman et al., 1995; Borum et al., 2005; Wicks et al., 2009). Therefore, to increase the forecasting ability of production models, the impacts of the sediment environment on the mechanisms that drive production and loss within these systems need to be accurately defined (Eldridge and Morse, 2000).

Summary

Seagrass production models are important tools to enhance our understanding of seagrass responses to stressors (Biber et al., 2004), further define the role of seagrass beds in coastal ecosystems (Baeta et al., 2011), and highlight areas where future research is needed (Best et al., 2001). Often seagrass production models provide information vital to well-informed management of coastal ecosystems that cannot be provided using field and laboratory data alone (JØrgensen, 1994; Best et al., 2001). While the predictive capability of models can be a powerful tool for research and ecosystem management, care needs to be taken to remember that predictions are limited based on the model's assumption of known interactions (Best et al., 2001).

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Cross-references

Ecological Modeling

SEA-LEVEL CHANGE AND COASTAL WETLANDS

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Synonyms

Falling sea level; Relative sea-level change; Rising sea level

Definitions

Coastal wetlands are wetlands within the zone of hydrologic influence of sea level.

The perimarine zone is an area where freshwater nontidal wetlands persist under the control of sea level.

Introduction

Sea-level-controlled wetlands comprise a wide variety of environments from salt marshes to freshwater marshes and swamps, fens, or barren salt flats, in a continuum of increasing elevation from a shoreline to the upland. In the case of the intertidal zone, the primary abiotic control on wetland structure and function is a combination of tidal inundation frequency, depth, and duration, known as hydroperiod (Brinson, 1993). Wetland environments below the highest astronomical tide experience direct tidal inundation, with decreasing frequency and duration as a function of increasing elevation within the tidal frame. For coastal wetlands landward, the water table is linked to the sea-level influence, which is an important control on the groundwater position that provides the waterlogged conditions necessary for their development (Hageman, 1969). The result of the interaction between hydrodynamics and elevation is a shore-parallel zonation of plants, where each zone tends to move both vertically and horizontally in response to changing sea level and associated stressors (Hayden et al., 1995).

Changes in relative sea level

While the globally averaged sea level has been rising from the last glacial maximum to the present, the relative height of the sea with respect to land (relative sea level) can vary from place to place due to local tectonic and hydrographic effects . As the mass of the continental ice melted, a huge weight was released from continental shelves, which rose by isostatic rebound of the land. In those areas where the ice load was the greatest and the largest rebound occurred, the land rose faster than the sea, the relative sea level decreased, the coast prograded, and new land emerged over the last 10,000 years (e.g., Fennoscandia, Finland, Labrador). In other areas, the coast initially receded from a rising sea until the relative sea level reached a transgressive maximum, after which the coast prograded, as the relative sea level decreased to its present elevation (e.g., the east coast of South America, Western Australia, and East China). Where the relative sea level fell rapidly, new land constantly emerged, the coastal wetlands continuum migrated seaward, and the Holocene estuarine environments became part of the terrestrial landscape. Where sea level rose, the Holocene estuaries were drowned and new wetlands formed landward.

A change in relative sea level produces an alteration in the ecological state of wetlands, and the different plant associations within the coastal wetlands continuum are expected to migrate in response to different hydrologic conditions. Coastal wetlands developing under a rising relative sea level during the Holocene have been largely studied along the eastern coast of North America, as well as marshes in northern Europe. The process of wetlands landward migration under a rising sea level was early described by Dutch geologists. Hageman (1969) termed the area where freshwater wetlands persist under the control of relative sea level as the perimarine zone, and he studied the evolution of freshwater swamps in the western Rhine/Meuse delta, in response to the rise in sea level during the Holocene. There are examples of sedimentary records (Waller, 1994; Kirby, 2001) showing that peat-forming perimarine wetlands accumulated deep layers of organic matter between about 6,000 and 2,000 years BP, and palynological analysis of these peat deposits showed sequences of salt marshes, reed swamps, fens, and woodland carr communities developing under a rising sea level, which maintained a near-surface water table (Waller et al., 1999). A similar model of transgression was described for the Virginia coast (Virginia, USA), a typical coastal barrier ecosystems extending along the seaward margin of the Delmarva Peninsula (Oertel et al., 1989). In this system, a sustained sea-level rise during the Holocene set up a similar sequence of state changes in wetlands along the mainland edge (Brinson et al., 1995). As transgression occurs, upland forest is replaced with high marsh, high marsh with low marsh, low marsh with mudflats, and mudflats with open water (Christian et al., 2000).

In contrast to these well-studied examples of continuous rising in relative sea level, little is known about wetland response in coastal environments that developed under different conditions after the last glacial age. Some examples from the Gulf of Bothnia describe a downward migration of plant zones in response to the continuous land uplift (Vartiainen, 1988; Ecke and Rydin, 2000) and the seaward expansion of pioneer plant communities (Zobel and Kont, 1992). A more complex dynamics characterizes wetland environments in the Atlantic coasts of southern South America, where the relative sea level reached a transgressive maximum during the Holocene (Cavallotto et al., 2004; Violante and Parker, 2004). In these systems, the late Holocene marine regression resulted in wide low-lying coastal landforms inherited from the former estuarine dynamics. These coastal environments are commonly occupied by perimarine wetlands, which undergo increasing inundation under the current rising trends in relative sea level. In the Bahía Blanca Estuary, Argentina, a rising sea level is a major cause of wetlands loss in elevated Holocene surfaces (Pratolongo et al., 2013), but the accelerated erosion of soft sediments is also the main source of suspended solids to the tidal sediment budget, allowing deposition and seaward expansion of low salt marshes (Pratolongo et al., 2010).

Accelerated sea-level rise

A major concern related to climate change is the recently accelerated sea-level rise associated with the melting of sea ice, land ice, and thermal expansion of the ocean (Webb III et al., 1993; Wigley and Raper, 1993). There has been considerable discussion as to how coastal wetlands will develop in the future under climate-enhanced sea-level rise (Reed, 1990; Simas et al., 2001). Early studies (Titus, 1987; Boorman et al., 1998) predicted the large-scale loss of coastal wetlands as a consequence of sea-level rise exceeding sediment supply. However, there is some evidence to suggest that, at some locations, the geomorphic response of salt marshes is not sediment limited. Many temperate salt marshes built from allochthonous sediment show a significant excess of vertical sediment accretion relative to sea-level rise (French, 2006; Stupples and Plater, 2007). In the Mississippi delta, accretion rates greater than 10 mm year^{-1} have been measured where there is sufficient sediment input from the river (Conner and Day Jr., 1991; Cahoon et al., 1995; Day et al., 2000), and mangroves in many estuaries in northern Australia tolerated sea-level rise of 8–10 mm per year in the early Holocene (Woodroffe, 1995). These accretion rates are higher than most projections and suggest that coastal wetlands can persist at a given location, in spite of high rates of sea-level rise, if there is sufficient mineral and organic soil formation.

Nevertheless, human activities alter the ability of wetlands to accrete both at local and regional scales, and enhanced sea-level rise has led to significant changes on coastal systems, mainly associated with salinity intrusion in estuaries and altered sediment transport. There are numerous examples of detrimental effects of accelerated sea-level rise on coastal wetlands around the world, including Chesapeake Bay, the Mississippi Delta, and other Atlantic estuaries in North America (Stevenson et al., 1985; Hackney and Cleary, 1987; Day et al., 2003; Day et al., 2007); Rhone, Ganges, Indus, Nile, and Ebro deltas ((Snedaker, 1984; Milliman et al., 1989; Stanley and Warne, 1993; Ibañez et al., 1999; Pont et al., 2002; Day et al., 2006); and Venice Lagoon (Pirazzoli, 1987; Day et al., 1999). Deltaic regions are particularly vulnerable to a relative sea-level rise because of rapid subsidence. Under this scenario, river sediment supply and human-induced changes in sedimentary fluxes are critical agents in shaping deltaic evolution. Dam construction and the increase in water demand for agriculture, industry, and tourist development have dramatically reduced the sediment load of rivers and are thought to be a major cause of deltaic degradation when coupled with subsidence (Stanley and Warne, 1993).

Vertical accretion and progradation resulting from sedimentation are not the only processes supporting wetland persistence. For coastal areas experiencing a relative rise in sea level, the different plant associations within the coastal wetlands continuum are expected to migrate landward, and the future extent of the wetland zone will depend on the combined effect of seaward vertical accretion, disturbance, and landward transgression (Christian et al., 2000). There are examples of coastal marshes and mangroves throughout the world, set against the land as a fringe parallel to the shore, that seem capable of responding to sea-level rise by moving inland, but there are also some exceptions. Wetlands growing on islands within estuaries have no land to migrate (Kearney and Stevenson, 1991; Wray et al., 1995). Similarly, the migration of wetlands inland may also be prevented in places where the landward slope is too steep or where people have built hard barriers landward of the wetlands. In these cases where transgression stalls, low sediment supply results in an eroding seaward margin, and wetland communities may disappear by erosion over time (Brinson et al., 1995).

Conclusions

Coastal wetlands have naturally evolved in response to global changes. Numerous studies show the resilience of coastal wetlands to natural disturbances. However, changes in sea level, coupled with anthropogenic changes on sediment loads, species introduction, nutrient enrichment, and other human alterations are likely to have a disproportionate impact on these systems.

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Cross-references

Coastal Wetlands
Estuarine Deltaic Wetlands
Managed Realignment
Salt Marsh Accretion

SECCHI DISK

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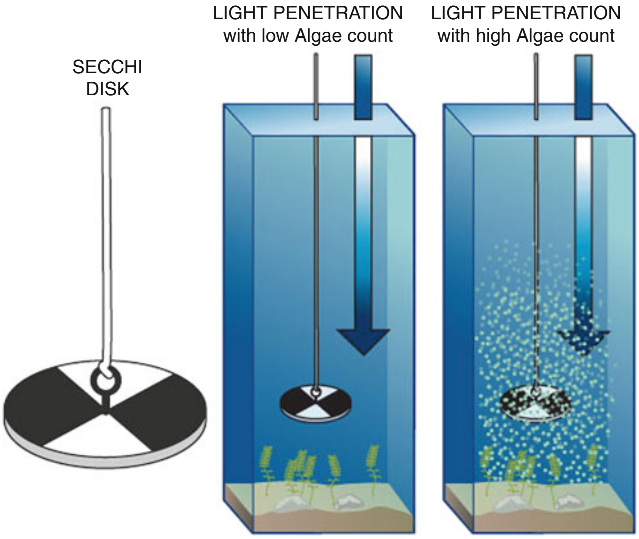
Definition

Secchi disk is defined as a simple, standard instrument used to measure water clarity in ponds, lakes, reservoirs, estuaries, and oceans (Hutchinson, 1957).

Description

It is an 8-in. (20 cm) diameter circular plate, evenly divided by black and white quadrants attached to a PVC pipe, dowel rod rope, or chain (Figure 1). The line contains marked measurements in inch or centimeter intervals on the rod, pipe, rope, or chain with permanent ink, paint, or clamps. The plate is lowered into the water of a lake or other water body, and the depth (Secchi depth) at which it is no longer visible from the surface is recorded. The rule of thumb is that light can penetrate to a depth of 1.7 times the Secchi disk depth.

Most disks used in freshwater bodies have alternating black and white quadrants, while disks used in marine environments are usually all-white. Early disks of the nineteenth century were all white; however, Whipple (1899) modified the original disk and "Whipple's" disk became the standard in freshwater environments. The all-black disk, developed and used in New Zealand, is used in shallow rivers and streams, because the black disk requires shallow water depths to measure water clarity.



Secchi Disk, Figure 1 Example of an alternating black and white Secchi disk commonly used in freshwater and marine water bodies to measure water clarity.

The Secchi disk is named after its inventor, Italian astronomer Angelo Secchi (1818–1878) (Cialdi and Secchi, 1965).

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Cross-references

Water Clarity
Water Quality

SECONDARY DUNE

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Synonyms

Back dune; Grey dune; Rear dune

Definition

Secondary dunes are also sometimes referred to as "rear dunes" and "back dunes" (http://www.islandbeachnj.org/Nature/physical/dunes.html) and "grey dunes" (http://people.uncw.edu/hosier/BIE/bieclschd/present/dneslkmorph.htm). According to Davies (1980), the terms "primary dune" and "secondary dune" are generic terms with relatively specific meanings. He defined "primary dunes" as dunes derived primarily from the beach, and he identified two types, namely, free dunes with vegetation unimportant (e.g., transverse ridges) and impeded dunes with vegetation important (e.g., foredunes). Davies' (1980, p. 157) "secondary dunes" are dunes derived from erosion of impeded primary dunes, and he documented two types, namely, (1) transgressive dunes (e.g., blowouts, parabolics, transgressive sheets) and (2) remnant dunes which are eroded remnants of vegetated primary dunes.

Other authors have defined secondary dunes in far less rigorous ways than did Davies (1980):

> The dune closest to the ocean is the primary dune, or foredune, followed by what are called secondary dunes or back dunes. (http://www.islandbeachnj.org/Nature/physical/dunes.html)
> A secondary dune is "created by modification of the primary dunes or by transfers of sand inland from the position of the primary dunes." (Psuty, 2008, p. 16)

> Secondary dunes (or rear dunes) are located farther inland (from the foredune or primary dune) and are not often directly exposed to marine influences. (http://www.crd.bc.ca/watersheds/ecosystems/coastalsanddunes.htm)

As Davies (1980) noted, his scheme is not entirely satisfactory. He stated that transgressive dunes are derived from the erosion of primary vegetated dunes, but later in their evolution may receive sand directly from the beach. We now know that transgressive dunefields and dune sheets may be initiated from erosion of foredunes or parabolic dunefields, but may be equally initiated from the backshore without any vegetated dunes ever being present (Hesp and Thom, 1990; Hesp and Walker, 2013). Transgressive dunes are therefore also "free primary dunes." It is common to now include Davies' (1980) "free primary dunes" (e.g., transverse ridges, barchans, etc.) under a broader classification of transgressive dunefields, so his class of free primary dunes completely overlaps with secondary transgressive dunes and sheets.

It could be argued that the terms "primary dunes" and "secondary dunes" should now be abandoned given we understand much more about coastal dune evolution, dynamics, and coastal dunefield classification (Hesp, 2002, 2011; Hesp and Walker, 2012).

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Cross-references

Back Dune
Foredune

SEDIMENT BUDGETS

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Synonyms

Sedimentary budget

Definition

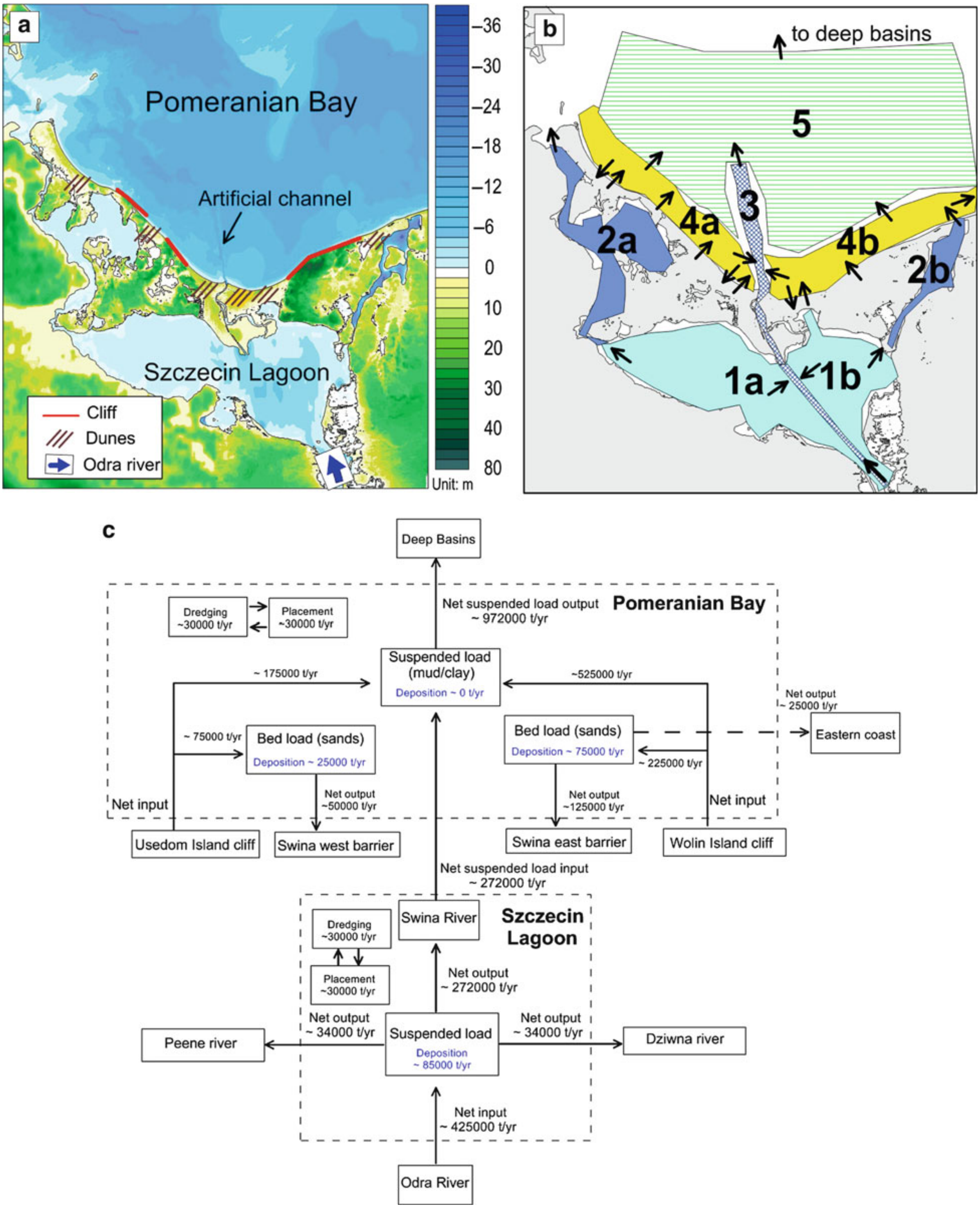
A sediment budget is an accounting of sediment volumes entering and exiting a particular region of an estuary or coast on the temporal scale of interest.

Introduction

Analysis of sediment budget is regularly applied in coastal and estuarine sciences and management studies to aid in understanding sediment sources, sinks, and transport pathways in a selected region of an estuary or coast within a predefined period of time. Depending on the temporal and spatial scales of interest, different hierarchies of knowledge complexity may be involved in accounting of sediment fluxes, sources, and sinks from different processes that give rise to additions and subtractions within the system (Slaymaker, 1997). For example, a coastal engineer may be interested in the physical processes (e.g., waves, currents) inducing sediment volume change at an inlet or a channel in a period of years or decades but may neglect the biological and chemical compositions of the sediment and their interactions with the environment. A marine biologist or chemist may be interested in the organic matter carried by mud that enters or exits a bay but probably has no interest in the volume change of gravels. A comprehensive study on the geomorphological history of a coastal system or a large lake, however, has to take into account all possible sources and sinks that contribute to the sediment budget. Thus, to construct the sediment budget for a specific estuary or coastal system on a temporal scale, possible sources, sinks, and transport processes must be identified prior and their temporal and spatial variations have to be integrated into analysis. The sediment budget of a coastal system or an estuary is usually constructed through the following procedures.

Procedures

1. Development of a "conceptual sediment budget" for the research area is recommended in the planning stage prior to making detailed calculations (Dolan et al., 1987). The conceptual budget serves as a qualitative model giving a regional perspective of possible sources, sinks, and transport processes, containing the effects of specific morphological units (e.g., shoals, inlets, cliffs, and anthropogenic structures). The conceptual model can be constructed in part by referring to existing sediment budgets developed for other sites with similar environmental settings (Rosati, 2005), and incorporating additional possible sediment sinks, sources, and transport pathways in the research area. An example of a conceptual sediment budget constructed based on existing data for a coastal system is shown in Figure 1c.
2. Collecting available datasets for the research area that are commonly used in sediment budget analysis. These may include (1) digital elevation models (DEM) covering both the terrestrial and subaqueous parts of the area



Sediment Budgets, Figure 1 (**a**) DEM and major morphological units in a river-lagoon-barrier system at the southern Baltic Sea, (**b**) cell division and sediment transport pathways in the system, and (**c**) constructed sediment budget of the system based on Emeis et al. (2002) and Zhang et al. (2013).

in different periods, including seabed mapping and profile measurements; (2) aerial photographs of the research area in different periods, which are especially helpful for indicating morphological changes and major transport pathways of the area; (3) records and data of anthropogenic activities in the area and related change of sediment properties and volumes; (4) records and data of natural hazards (e.g., storms, floods) occurring in the area within the time span of interest; (5) meteorological conditions of the area (e.g., wind and wave monitoring data) and possible climate change impacts (e.g., sea level change data); (6) tectonic movement map of the area; (7) map of sediment grain-size distribution on the seabed; (8) measurements of suspended sediment concentration and discharge in representative periods (e.g., flood and ebb tides, dry and wet seasons); and (9) estimates of waves, currents, and sediment transport aided by numerical models.

3. Dividing the research area into a series of cells. Each cell acts as a control unit with clearly defined boundaries. Sediment budget is cell-dependent, and calculation of sediment volume change is performed only at the boundaries of each cell. Cells can be defined by geological or morphological controls, available data resolution, coastal structures, and knowledge of the site (Rosati, 2005). Depending on the specific sources, sinks, and transport processes in the research area, cells with different sizes can be assigned. Sub-cells can also be defined within a cell to better measure the sediment budget of a region with varying rates of accretion and erosion. A well-known example is given by Bowen and Inman (1966) who introduced the concept of littoral cells. A littoral cell is usually a zone parallel to the shoreline, bounded by the foot of the foredune or cliff in the landward direction and depth-closure point in the shoreward direction. Changes of the sediment volume in a littoral cell directly induce changes in the coastline, and ideally they are defined to minimize longshore sediment exchange with adjacent cells, for example, setting the lateral boundary of a littoral cell at the nodal points where the net longshore transport rate is zero or defining a pocket beach bounded by rocky headlands, which is able to conserve its majority of sediment, as a littoral cell. An example of cell division of an estuary and its adjacent coast is shown in Figure 1a, b.
4. Defining sediment transport pathways at the cell boundaries. Sediment transport pathways specify the sediment transfer direction and corresponding processes between adjacent cells. The transport pathways can be estimated through knowledge of the processes (natural and anthropogenic) occurring at the site. Interpretation of aerial photographs, trend analysis of sediment grain-size distribution, and particle tracking also aid to define sediment transport pathways. Figure 1b shows possible sediment transport pathways for an estuary and its adjacent coast.
5. Calculating the volume of sediment transfer in each transport pathway. A general equation for the sediment volume change (ΔV) of a cell is expressed by:

$$\Delta V = \sum Q_{\mathrm{source}} - \sum Q_{\mathrm{sink}} + I - O \tag{1}$$

where Q_{source} and Q_{sink} represent the amounts of natural sources (i.e., input) and sinks (i.e., output) transferred at the cell boundary, respectively. I and O are the amounts of artificial sediment input (e.g., beach nourishment) and output (e.g., dredging) transferred at the cell boundary, respectively. The quantity of sediment volume transfer on each transport pathway should be evaluated based on the acquired data. For example, comparison among aerial photographs in different periods gives the detailed information on the rates of coastline change. Field measurements on currents, waves, and sediment concentrations help to quantify the typical rates of sediment transport caused by natural processes. Analysis of extreme events provides information on the frequency and magnitude of consequent sediment transport. Comparison among DEMs, coastal profiles, or aerial photographs provides information on calibration and validation of the estimated quantities in (1).

6. Estimation of uncertainty in the calculated sediment budget. One should consider that uncertainty always exists in the calculated sediment budget. Uncertainty of a sediment budget mainly comes from two sources: (1) measurement error and (2) true uncertainty due to temporal and spatial variability of the transport processes (Rosati, 2005). Measurement error contains the error made during measurement processes (e.g., improper positioning of the instruments) and the error in the measured signals (e.g., noises). Net sediment flux across an estuary mouth or an inlet is difficult to measure, particularly when the sediment flux on the ebb and flood tides are high. In estuaries, sources and sinks can be expressed as rates of exchange in a tidal cycle. However, many of the exchanges are not at steady rates, for example, sediment exchange in stochastic extreme events (e.g., storms, floods) can be several orders of magnitude higher than in normal conditions. Another source of uncertainty comes from numerical models which are applied. For example, littoral drift is normally estimated on the basis of standard equations (e.g., the Coastal Engineering Research Center (CERC) (USACE, 1984) and Kamphuis (Kamphuis, 1991) methods) or more complicated models (e.g., Zhang et al., 2013). However, different models may yield quite different results on the rate of longshore sediment transport even though they are based on the same boundary inputs. Thus, the estimated littoral drift is often biased and should be calibrated by field measurements. Impacts of unknowns

(e.g., sediment porosity) and subordinate processes that are not taken into account in calculating the sediment budget also contribute to the uncertainty.

The degree to which one process affects another and the contribution of a transport process to the sediment budget both depend on the magnitude and frequency of recurrence of these processes at a site (Dietrich et al., 1982). A feasible way to better evaluate the uncertainty is to develop a scheme which presents the probability distributions of all possible sources and sinks in the research area.

Summary

Sediment budget analysis is useful in estuarine and coastal studies. A comprehensive understanding of the sediment source-to-sink transport and possible transport pathways within a system can be gained through a detailed budget analysis. Due to the complexities of sediment dynamics, much progress is still needed to reduce the uncertainty (e.g., improving the quality of field measurements and knowledge of sediment transport by multi-scale processes) for a better quantification of sediment budgets in coastal and estuarine environments.

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Cross-references

Sediment Grain Size

SEDIMENT COMPACTION

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Synonyms

Sediment dewatering; Sediment lithification

Definition

Clay, silt, gyttja, and peat are sediments that contain substantial amounts of water at the time of deposition. As the sediments continue to accumulate through time, they undergo dewatering which leads to compaction.

Description

The transition from particles in suspension to their accumulation in bottom sediments implies the loss of water so that the particles hold together by cohesion of friction. The process of dewatering continues in the upper part of the sediment column until some sort of equilibrium is reached (Mörner, 2010; Brian et al., 2012). This is the case with clay, silt, and gyttja (organic matter). Peat is susceptible to compaction (e.g., Jelgersma, 1961). Loading generates sediment compaction (e.g., when heavy harbor construction rests on soft sediments). Many tide gauges are located on such construction, which leads to site-specific subsidence (e.g., Mörner, 2010). Dewatering and compaction are parts of the process of lithification. Artificial water withdrawal may generate substantial sediment compaction (e.g., in the Bangkok region and in the Nigita area in Japan). Compaction is a serious problem in the reconstruction of sea-level changes adding a factor of local to site-specific subsidence which may be hard to define.

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Cross-references

Estuarine Sediment Composition
Sediment Erosion
Sediment Grain Size

SEDIMENT EROSION

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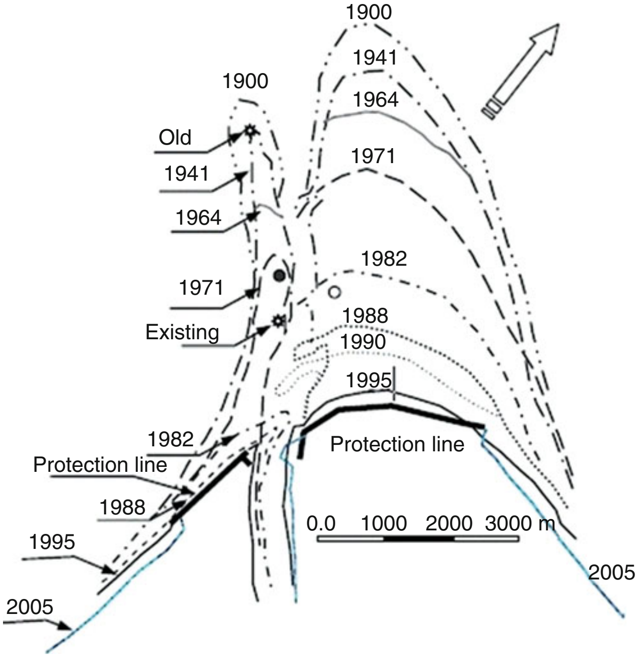
Definition

Sediment erosion is the process of detachment and transport of sediments by water, wind, ice, and gravity (Kumar and Ramachandra, 2003).

Description

Estuarine erosion takes place by the wearing away of shoreline and bottom sediments (Rogers and Skrabal, 2011). On an estuarine beach, sediment erosion occurs primarily by wave action, tidal currents, littoral currents, and deflation (CERC, 1984).

Erosion is a natural geological process, and the rate of shoreline and bottom sediment erosion will vary from place to place (Rogers and Skrabal, 2011). It can be significantly modified by human activities (Striebig, 1999). The severe erosion of the Nile River Rosetta Estuary, Egypt (Figure 1), after establishment of Aswan High Dam, is a good example of erosion due to anthropogenic activities (Fanos, 1995; Stanley and Warne, 1998). Coastal defense structures, particularly breakwaters, were constructed along the two sides of the Rosetta Promontory on land (Fanos et al., 1995). These seawalls were 5.3 m above mean sea level and 8 m deep. Mediterranean water has been in contact with these seawalls since 1995. Erosion did not stop but continues vertically to the bottom sediments, with rates from 0.1 to 0.5 m/year in front of the seawalls (Frihy et al., 2008; El-Gamal, 2012).



Sediment Erosion, Figure 1 Erosion of the Nile River Rosetta Estuary after construction of the Aswan High Dam (Fanos, 1995).

The actual erosion rate within an area may vary within estuarine systems and over time, depending upon individual site conditions and the frequency of storms or other causes of erosion (Rogers and Skrabal, 2011). Breaking waves only several centimeters in height have the energy to move sand and other sediments both offshore into deeper water and alongshore. Once coastal sediments are in motion, they are often redistributed based on grain size and density (Rogers and Skrabal, 2011). High-density materials tend to concentrate in areas of beach erosion, whereas minerals of lower density and coarse size are selectively entrained by waves and currents and preferentially transported to zones of beach accretion where they are deposited (Frihy et al., 1995; El-Gamal and Saleh, 2012).

El-Gamal (2012) summarized different indicators for recognizing where erosion and accretion take place in beach profiles. Rogers and Skrabal (2011) grouped the types of structures and methods for managing or controlling estuarine basin erosion from the shoreline to bottom areas. These include:

- Land management (advance planning of building locations and other development activities)
- Vegetation (e.g., marsh plants)
- Beach fill or nourishment (addition of sand to a beach to compensate for expected or realized losses)
- Shoreline hardening (e.g., bulkheads and seawalls)
- Sand traps (e.g., groins and breakwaters)

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Cross-references

Sediment Transport

SEDIMENT GRAIN SIZE

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Synonyms

Grain size analysis; Grain size distribution; Sediment particles size; Texture

Definition

Texture refers to the general physical appearance of the sediment.

Grain size is the average size of the grains in a sediment sample. It is also known as the particle size.

Sand consists of grains of particle size ranging from 0.0625 to 2 mm (0.002–0.08 in.). It pertains to particles that lie between silt and granules on the Wentworth scale of grain size. Sand size class ranges from −1.0 to 4.0 (phi).

Silt consists of grains of particle size ranging from 0.008 to 0.0625 mm (0.0003–0.002 in.). It is intermediate in size between sand and clay. Silt size class ranges from 4.0 to 8.0 (phi).

Clay consists of grains of particle size between silt and colloid. These include any of the various hydrous aluminum silicate minerals that are plastic, are expansive, and have ion-exchange capacities. Clay size class ranges from 8 (phi) and onwards.

Introduction

Sediment is made up of loose particles of sand, silt, and clay. Particle size refers to the diameter of individual grains of sediment. It is a fundamental descriptive measure of sediments from any environment. Grain size analysis of estuarine sediments is required to study the trends in surface processes related to dynamic conditions of transportation and deposition.

Measures of the grain size distribution

The nature of grain size distribution in sediments of estuarine or any environment can be described on the basis of five specific parameters. The graphic mean size is an arithmetic average of a series of diameter values. The median diameter is the 50th percentile diameter of a cumulative frequency curve drawn on arithmetic probability paper. Standard deviation is expressed as measures of dispersion (sorting) of sediments, and it is the square root of the arithmetic average of the squares of all the deviations from the mean size value of a series of observations. Skewness measures the asymmetry of the grain size distribution. Grain size distribution is skewed when the mean deviates from the median. Skewness of the sediments for symmetrical grain size distribution is zero. Skewness becomes negative when the grain size is skewed toward smaller phi value, and it is positive when skewed toward higher phi value. Kurtosis is the condition of peakedness or flatness of the graphic representation of a statistical distribution.

Expression of sediment grain size

Estuarine waters transport a wide range of sediments varying in size from 2 μm (0.002 mm) to more than 4 mm, but finer sizes dominate most estuaries. A few estuaries transport sand (>62 μm), gravel, and larger sediments.

Sediment grain size is measured in metric units as propounded by Wentworth (1932). It was expressed as phi (ϕ) by Krumbein (1938), since the logarithmic diameter has more significance in a discussion of the statistical relations of sediments. Sediment grain size in phi (ϕ) is expressed as the negative logarithm to the base 2 of the sediment particle diameter in mm. Thus sediment grain size is expressed as follows:

$$\phi = -\log_2 \varepsilon$$

where ϕ is the sediment grain size and ε is the negative numerical value of the diameter. ε is equal to 2, 1, ½, ¼, etc., whereas ϕ is equal to −1, 0, +1, +2, etc. Thus ϕ increases with decreasing diameter.

Sand is the particle size of 0.0625–2 mm (0.002–0.08 in.). It pertains to particles that lie between silt and granules on the Wentworth scale of grain size. Sand size class ranges from −1.0 to 4.0 (phi). Silt is a particle size of 0.008–0.0625 mm (0.0003–0.002 in.). It is intermediate in size between sand and clay. Silt size class ranges from 4.0 and 8.0 (phi). Clay is a particle size between silt and colloid. Any of the various hydrous aluminum silicate minerals are plastic, are expansive, and have ion-exchange capacities. Clay size class ranges from 8 (phi) to higher.

Sampling of estuarine sediments

Utmost care is needed in sampling estuarine sediments because the grain size analyses are sensitive to the manner in which the original samples are collected, handled, and preserved. Introduction of any foreign particle into the sample through improper care, cleaning of equipment, or processing can alter the texture. Estuarine landforms such as point bars, river mouth bars, tidal shoals, major tidal inlets, and upstream and downstream of the rivers are ideal sites for the collection of samples. Data regarding tide, current, waves, depth, turbidity, etc. are also collected during sampling of estuarine sediment for size analysis. An instrumental tripod ALICE fitted with various sensors is used to collect data regarding the above physical parameters. Numerical models are used for data interpretation.

Statistical analysis of grain size

The texture of muddy sediments in estuaries is examined by mechanical analysis following the sieving-cum-pipetting method. Sand and gravel fractions are determined by sieve analysis (Krumbein and Pettijohn, 1938) using sieves of different mesh sizes marked as ASTM (American Society for Testing and Materials). The statistical size parameters are calculated using the formula of Folk and Ward (1957) from the cumulative curves drawn on arithmetic probability paper. Subsequently, rapid sediment analyzers (RSA) propounded by Zeigler et al. (1960) and Schlee (1966) and electro-resistance multichannel particle-size analyzers (EMPSA) are introduced for automated analysis and calculation of statistical parameters of sediments. Contemporaneous with them, Kane and Hubert (1962) and Schlee and Webster (1967) developed Formula Translation (FORTRAN) programs for textural analysis of sediment particle parameters. Gradually, Algorithmic Language (ALGOL) by Jones and Simpkin (1970), Beginner's All-Purpose Symbolic Instruction Code (BASIC) by Sawyer (1977), and handheld calculators by Benson (1981) were programmed for statistical grain size computations. About the same times, many workers (Muerdter et al., 1981; Poppe et al., 1985) and organization (Coulter Electronics Inc., 1989) introduced hardware and software packages for electro-resistance multichannel particle-size analysis. Introduction of computer-driven, integrated particle-size analysis instruments fitted with settling tubes (Zeigler et al., 1964; Rigler et al., 1981) automated and modernized sediment grain size analysis. The settling tube, also called rapid sediment analyzer design based on using the pressure differential between two columns of water that have a common head, provides for efficient analysis of sand-sized material by setting the grains where results are relayed to a personal computer associated with data acquisition software drivers (Syvitski et al., 1991). A computer program called GRADISTAT (Blott and Pye, 2001) has been written for the rapid statistical analysis of size data from any standard measuring technique. The program runs with a Microsoft Excel package. It is very useful and produces a range of graphical representations, including frequency curves and plots.

Cumulative curves

Cumulative curves plotted on arithmetic probability paper represent grain size distributions of different subpopulations which have a lognormal distribution depicting different modes of transportation of sediments (Visher, 1969). Sediment grain size is determined from grain size curves drawn on log-probability plots. It helps in the interpretation of separate populations of estuarine sediments. Three different methods of plotting are considered for grain size distribution, including grain size with frequency percent, cumulative frequency percent, and the log-probability cumulative frequency percent. Log-probability cumulative frequency curves are the most accepted methods used by sedimentologists in assessing depositional environments of estuaries. In each log-probability curve, there are at least four control points, i.e., four separate lognormal populations, where each population is truncated and the former one joined with the latter one to make a single grain size distribution. Each lognormal population is composed of different mean and standard deviation values.

Grain size characteristics of different estuarine landforms

There is an interrelationship between grain size characteristics and the depositional pattern in a tide-dominated estuarine environment. Interpretation of the grain size frequency curve is based upon the pattern of curves and splitting of each curve into segments separated by the marked breaks and inflection. Sediments from different geomorphological areas such as point bar, mid-channel bar, swash bar, river bank, and areas of other morpho-ecological interests may be considered for grain size analysis.

Texture of mudflat sediments

The graphic mean size of surficial and subsurface samples lies within the silt fraction with moderately well to poor sorting. Sediments of mudflat samples show a positive skewness and reflect infiltration of suspended clay from tidal standstill through the pore spaces of the dominating silt and subordinate sand populations. Sediments of the mudflats traversing the creek bottom, however, exhibit negative skewness. This is because of the mixing of a greater proportion of sand fraction with the dominant silt population. The removal of clay with flowing creek water leaves the creek bottom with more sands compared to other places of the mudflat, and this leads to a negative skewness of the distribution patterns. Cumulative curves drawn from the mudflat sediments reveal close similarity in pattern. The same is true for the creek bottom sediments when considered separately. The prominent breaks in the cumulative curves reflect changes in the mode of transport of suspended particles.

Texture of sand flats and silt flats

The sediments of sand flats consist of well-sorted 95 % fine to very fine sands in comparison to silt flat sediments having 10 % sand, 95 % silt, and 5 % clay. Cumulative curves for the sand flat sediments are very similar in pattern and differ much from the silt flat sediments.

Texture of tidal shoal sediments

Tidal shoals, in general, show greater accumulation of mud in the upstream portion and sand in the downstream stretches. This characteristic depositional behavior of sediments favors tidal accumulation rather than its formation from unidirectional flow from the upper to the lower stretches. The downstream accumulation of sand suggests the influence of flood flow, whereas the upstream mud is the result of a standstill during the high and low tides.

The sediments are generally very well-sorted to medium-sorted fine silt in nature. Samples are slightly negative to slightly positively skewed (SK_1) and are platykurtic in nature. Inflections generally at two to three truncation points are a statistical fact for all the samples. All the cumulative curves are mostly nonlinear, showing close resemblance in pattern with each other.

Texture of point bar sediments

Texturally, the sediments are generally composed of 95 % silt, with subequal proportions of fine sand and clay forming the remaining 5 %. Sediments are mostly well sorted to moderately sorted, with graphic mean size belonging mostly to the silt fraction. Most of the sediments show a slight tendency toward negative skewness. The cumulative curves mainly exhibit higher peakedness, with K_G values often close to 1.0.

Sandy sediments of the point bars show a completely different pattern from that of sandy silty sediments. The cumulative curves for these sediments show a nonlinear pattern and are perfectly comparable to that of the sandy point bar sediments. These sediments display two major inflections at 2.25 phi and 3.25 phi, respectively. These inflections divide the curves into three subpopulations as rolling, siltation, and suspension, respectively. The saltation population constitutes about 75 % of the materials, the rest being deposited by either rolling or suspension. Silty sediments having range from 4 to 9 phi show exactly the same pattern where the three subpopulations can be well recognized. In these curves, the inflections take place at 6–7 phi, respectively. The central saltation population constitutes about 75 %. These sediments do reflect their deposition from tractive movements of water in the estuarine flow condition.

Texture of marsh sediments

The marsh sediments are composed of 90 % silt with subequal proportions of fine sands and clays forming the rest. The marsh sediments are mostly moderately well sorted to moderately sorted, with the graphic mean size belonging to the silt fraction. Most samples exhibit a slight tendency of negative skewness. The negative skewness in the marshy region of the estuaries perhaps indicates trapping of larger bed-load particles by the marsh vegetation. The cumulative curves primarily show a high peakedness with greater K_G values. The statistical size parameters of the sediment samples are very much analogous to lagoon or distal shelf sediments.

Texture of mangrove swamp sediments

Tidal flats are mostly siliciclastic. The sediments are generally silty to sandy. All the cumulative curves are non-lognormal. The silty sediments are extremely zigzag and reflect deposition out of ebb-flood cycles. The statistical size parameters show that the sediments are mainly well sorted to moderately sorted, with graphic mean size belonging to the silt fraction. Most sediment samples show a slight tendency of negative skewness. The cumulative curves largely show high peakedness, with K_G values often greater than 1.0. The cumulative curves for the sandy sediments exhibit three distinct subpopulations (i.e., rolling, saltation, and suspension subpopulations). The cumulative curves for the silty sediments, however, show more intricate zigzag patterns reflecting ebb-flood cycles.

Texture of dune sands

The grain size of dune sands generally shows lognormal distribution with very fine sand and well-sorted sediments. The graphic mean size ranges between 2.5 and 3.5 phi. The graphic standard deviation indicates well-sorted finer sand particles. The sorting value reflects reworked intertidal zone sediments through wind transportation cycles. Dune sediments exhibit a slightly positive skewness because of admixture of very small quantity of finer suspended particles to the lognormal saltation population. Kurtosis of sediments is very close to 1.0 indicating lognormal distribution with mesokurtic characteristics.

Textural sensitiveness

There is rhythmicity in the nature of deposition, which perhaps indicates the depositional pulses for the ebb and flood flows through rivers. The inflection points in the curves between successive ebb-flood cycles are marked at 2.5–3.0 phi, 3.5 phi, 5–6 phi, and 8 phi sizes, respectively. Such rhythmicity in the nature of depositional behavior of tidal sediments is supposed to be highly process responsive.

Sandy sediments (ranging from 1.5 to 4 phi) from the point bars and mid-channel bars of estuaries display a completely different pattern from that of sandy silt sediments. The cumulative curves for these sediments exhibit a nonlinear pattern and are comparable to that of the other point bar sediments. These sediments have two major inflections, one at 2.25 phi and the other at 3.0–3.5 phi. These inflections divide the curves into three subpopulations as rolling, saltation, and suspension, respectively. The saltation population generally constitutes about 75 % of the total material, the rest being deposited by either rolling or suspension. Silty sediments having a range from 4 to 9 phi from similar areas also show exactly the same pattern where the three abovementioned subpopulations can be well recognized. In these curves, the inflections take place at 6–7 phi, respectively. The central saltation population constitutes about 70 %. These sediments reflect their deposition from tractive movements of water in a unidirectional flow condition.

Summary

Sediment grain size is considered one of the most important tools for the interpretation of depositional environments in estuaries (Das, 2009). Sorting indicates the process of modification of the sediments, whereas graphic mean size reflects the environment of sediment accumulation. Rigorous flow transports the sediments in the depositional environment causing poor sorting of sediment particles. Sediments are skewed in selective transportation, and a particular sediment population is characterized by inclusive graphic kurtosis (Davis, 1983).

Grain size of estuarine sediments reflects the nature of source sediments and their hydrodynamic condition of deposition. Generally erosion dominates along the seaward reach of the estuary, with high wave energy and deposition predominating in the landward reaches of relatively quieter environment. Thus finer muddy sediments are deposited on the estuarine banks and flanks of the mid-channel bars and point bars, with low depositional energy.

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Cross-references

Sediment Sorting
Sediment Transport

SEDIMENT RESUSPENSION

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Definition

Sediment resuspension is the suspension and redistribution of previously deposited sediment particles in the water column due to hydrodynamic forcing.

Sediment suspension is the mobilization and entrainment of sediment particles from the bed due to hydrodynamic forcing.

Introduction

Sediment resuspension plays a critical role in estuarine sediment budgets. Sediment fluxes in an estuary can be contributed from both marine and terrestrial origins. Resuspension of marine sediments, mainly sand, by nearshore waves, currents, and wave-induced currents, shapes coastal landforms (Dean and Dalrymple, 2002). Often, marine sand can also enter (or reenter) an estuary facilitated by tidal asymmetry and estuarine circulation (MacCready and Geyer, 2010). A significant amount of terrestrial sediment is fine-grained, such as clay and silt. Fine-grained sediments become cohesive in estuaries and form floc aggregates (also called mud) through flocculation (Winterwerp and van Kesteren, 2004). Flocs are vehicles of organic carbon, nutrients, and pollutants, which further leads to many ecological and geochemical outcomes in the water column and benthic boundary layer (Santschi et al., 2005). When terrestrial sediments are delivered to estuaries or river mouths, significant trapping and deposition occur due to diminishing flow intensity, estuarine stratification, and flocculation (Wright, 1977; Geyer et al., 2004). Hence, sediment resuspension becomes the key process to further deliver sediments offshore before the journey of sediment source to sink can be completed (Wright and Nittrouer, 1995). Although sediment resuspension is driven by the overlaying hydrodynamics, resuspension also leads to a variety of seabed characteristics, such as bedform and fluid mud, which in turn determines the bottom friction (dissipation) experienced by the overlaying hydrodynamics.

Mechanisms of sediment resuspension

Sediment resuspension is generally driven by bottom boundary layer flow, which is a layer of sharp transition of flow velocity from nearly zero at the bed to a large magnitude of overlaying hydrodynamics. Hence, the flow shear in a bottom boundary layer, defined here as the vertical gradient of streamwise flow velocity, is significant, and it is the main ingredient of turbulence production and dispersion of solutes (Pope, 2000). Consequently, surficial bed sediments are mobilized and suspended in the bottom boundary layer. Boundary layer flow and sediment resuspension can be driven by tidal currents, river outflows, surface waves, and even internal waves. Moreover, anthropogenic causes such as dredging and ship waves can also cause sediment resuspension. Once sediments are suspended in the bottom boundary layer, depending on the bed slope and the characteristics of suspension (see next section), downslope gravitational force may also play a critical role to transport sediment in the downslope direction.

Resuspension processes

Although turbulence is the key mechanism of sediment suspension, when a significant amount of fine sediments is suspended in the bottom boundary layer, flow turbulence can be attenuated through sediment-induced density stratification. This is typically observed through the formation of the “lutocline,” a sharp negative gradient of suspended sediment concentration at some elevation above the bed. Such turbulence-sediment interaction leads to several critical processes. In moderate sediment concentration (or relatively intense overlaying flow), the lutocline separates the lower turbulent boundary layer from the upper nonturbulent layer. Hence, sediments are mostly accumulated below the lutocline and may establish sufficiently large buoyancy anomaly to drive offshore-directed gravity flow (Traykovski et al., 2000; Wright and Friedrichs, 2006). Because flow turbulence eventually leads to energy dissipation through energy cascade (Pope, 2000), when turbulence is attenuated by the presence of sediments, mean flow kinetic energy increases. This phenomenon is called drag reduction. During AMASSEDS (A Multidisciplinary Amazon Shelf Sediment Study; Nittrouer et al., 1991), a significant reduction of bottom drag coefficient is observed as tidal currents propagate over a muddy bed (Beardsley et al., 1995), where a thick layer of mud up to several meters is present (Trowbridge and Kineke, 1994). The existence of the lutocline also implies significantly suppressed mixing of other solutes and hence effects on benthic boundary layer. Since the presence of sediment can attenuate turbulence, there exists a carrying capacity for both current-dominant and wave-dominant sediment resuspension (Winterwerp, 2001; Ozdemir et al., 2011), and that for a given flow intensity and sediment characteristics (e.g., settling velocity), the flow can only sustain a maximum amount of sediment load. When the carrying capacity is exceeded (e.g., flow intensity decreases when approaching slack water), turbulence in the boundary layer is significantly suppressed and catastrophic sediment settling occurs. Also facilitated by hindered settling effect, a thick layer of concentrated sediments is accumulated near the bed, called fluid mud, and experiences a slow consolidation process (Mehta, 1991). The formation of fluid mud and a laminarized boundary layer give rise to a greatly enhanced bulk flow viscosity near the bed, which appears to also cause a large surface wave dissipation rate (Sahin et al., 2011).

In the intertidal environments, large sediment resuspension at the intratidal time scale is often observed at very shallow water depths near the land-water interface, called turbid tidal edge (e.g., Christie and Dyer, 1998; Nowacki and Ogston, 2013). On the daily and fortnightly time scale, a significant amount of sediment is exchanged between the tidal channel and adjacent flats. The net exchange is highly dependent on seasonal variability due to vegetation. For example, observations at the intertidal mudflats of Willapa Bay (USA) show that during the winter period, flats are not vegetated and channels are filled with thick layers of muddy deposits delivered from the flats during ebb flow. On the contrary, with only limited vegetation during the summer period, sediments appear to be trapped on the flats, and hence the channels are observed to be free of mud (Boldt et al., 2013).

Summary

New sensor technology has provided a wealth of insights into the sediment-laden bottom boundary layer regarding fluid-sediment interactions and sediment properties that are essential to the understanding and modeling of sediment resuspension (Mikkelsen et al., 2004; Traykovski et al., 2007). In recent years, numerical simulations based on two-phase flow principles are capable of resolving most of the three-dimensional turbulence-sediment interactions; they also reveal new insights on the mechanism of sediment resuspension (Cantero et al., 2009; Ozdemir et al., 2010). Several critical aspects of sediment resuspension warrant future studies. A robust flocculation module needs to be incorporated into models for sediment resuspension. On the same note, appropriate parameterization of erosion flux for cohesive sediment bed capturing consolidation and characteristic of aggregates is necessary

(Sanford, 2008; Winterwerp et al., 2012). Moreover, the role of vegetation on sediment resuspension needs to the better quantified (Nepf, 2012).

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Cross-references

Sediment Budgets
Sediment Erosion
Sediment Grain Size
Sediment Transport
Tidal Hydrodynamics
Wave-Driven Sediment Resuspension

SEDIMENT SORTING

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Definition

Sediment sorting is the degree of dispersion of a grain-size distribution around a central value (mean, median, or mode).

Description

Sorting can reflect both sediment source and/or transport, with aeolian-transported sediments being among the best sorted and glacial sediments being among the poorest sorted. The measurement of the degree of sorting of a grain-size distribution can be given by any of the statistical dispersion measurements; the standard deviation is the most common. The kurtosis (or "peakedness") has also been widely used by investigators as a sediment sorting parameter, especially between 1960 and 1980.

One of the most frequently used sorting parameters is the "inclusive graphic standard deviation" proposed by Folk and Ward (1957), in which the 68 % and 90 % of the median value have been employed to define sorting criteria: very well sorted, well sorted, moderately well sorted, moderately sorted, poorly sorted, very poorly sorted, and extremely poorly sorted.

After Folk and Ward's seminal paper, several other works used grain-size parameters to delineate between environments (Sahu, 1964). Another approach was the use of grain-size parameters to determine the net transport in beach, estuarine, and shelf environments. In all of the papers published on this subject, sediment sorting was shown to play a major role in the determination of the direction of transport, since it is assumed that sorting is always better towards the direction of transport (McLaren and Bowles, 1985; Gao and Collins, 1992).

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Cross-references

Sediment Grain Size

SEDIMENT TOXICITY

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Definition

Sediment toxicity is a measure of the negative impact of contaminated sediments on aquatic organisms.

Background

Contaminated sediments can potentially be detrimental to aquatic organisms, both benthic and pelagic, and therefore have negative impacts across aquatic ecosystems. Contaminants may be directly toxic to aquatic life or can be a source of contaminants for bioaccumulation in the food chain (USEPA, 1994a). The concentration of contaminants in sediments can be several orders of magnitude greater than in the overlying water, and therefore measurements of water quality may differ greatly from sediment quality (USEPA, 1994a). The bioavailability of a contaminant is sensitive to local environmental variables, including sediment geochemistry, pH, and oxygen concentration.

Chemical, biological, and ecological methods of quantifying sediment quality have been developed; however, each method has its shortcomings (Chapman, 1989). For example, assessing toxicity by measuring individual chemicals in sediments may miss unmeasured chemical compounds or may not account for changes in bioavailability for different sediment types. In addition, the additive effect of mixtures of chemicals further complicates the prediction of sediment toxicity based on chemical data. To determine whether contaminants in sediments are harmful to benthic organisms, the EPA has developed methods that measure the survival after 10-day incubations of a freshwater amphipod (*Hyalella azteca*) or midge (*Chironomus tentans*) or an estuarine or marine amphipod (*Ampelisca abdita*, *Eohaustorius estuarius*, *Leptocheirus plumulosus*, and *Rhepoxynius abronius*) (USEPA, 1994a, b). Methods to determine sublethal effects, including effects on reproduction and growth, have also been developed (USEPA, 2000, 2001). To effectively determine the magnitude and extent of sediment contamination, data from several different methods must be integrated.

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Cross-references

Halogenated Hydrocarbons
Marsh Sediment Toxicity
Oil Pollution
Trace Metals in Estuaries

SEDIMENT TRANSPORT

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Definition

The simplest definition of sediment transport is the transport of granular particles by fluids. The main agents by which sedimentary materials are moved include gravity (gravity transport), river and stream flow, ice, wind, and estuarine and ocean currents. Running water and wind are the most widespread transporting agents. In both cases, three mechanisms operate, although the particle size of the transported material is very different, owing to the differences in density and viscosity of air and water. The three processes are rolling or traction, in which the particle moves along a sedimentary bed but is too heavy to be lifted from it; saltation; and suspension, in which particles remain permanently above the bed, sustained there by the turbulent flow of the air or water (U.S. Army Corps of Engineers, 2003).

Importance of sediment transport

Sediment transport has been studied for centuries and remains a challenging area of research for earth and marine scientists. The general term of "sediment transport" includes a number of environmental processes that take place at a wide range of spatial and temporal scales. The full understanding of sediment transport is fundamental to assessing a range of heterogeneous geological, engineering, and environmental processes. Over the recent years, the development of sediment transport research was transformed from descriptions of simple empirical\phenomena to more complex numerical models in which the flow and the resulting sediment transport are detailed.

The study of sediment transport processes includes the movement of particles, rocks, and other earth materials by various processes. Transport is driven by gravity effects and by friction effects with the air or the fluid containing the sediment. Sediment transport due to fluid motion occurs in rivers, lakes, estuaries, seas, and other bodies of water due to currents and tides, in glaciers as they flow, and on terrestrial surfaces under the influence of wind. Sediment transport due only to gravity can occur on sloping surfaces in general, including hill slopes, scarps, cliffs, and the continental shelf – continental slope boundary.

Sediment transport is usually divided into three types: bed load, saltation, and suspension. Bed-load transport is defined as the type of transport where sediment grains roll or slide along the bed. Saltation transport is defined as the type of transport where single grains jump over the bed at a length proportional to their diameter, periodically losing contact with the bed. Sediment is suspended when the flux is intense enough such that the sediment grains are suspended over the bed.

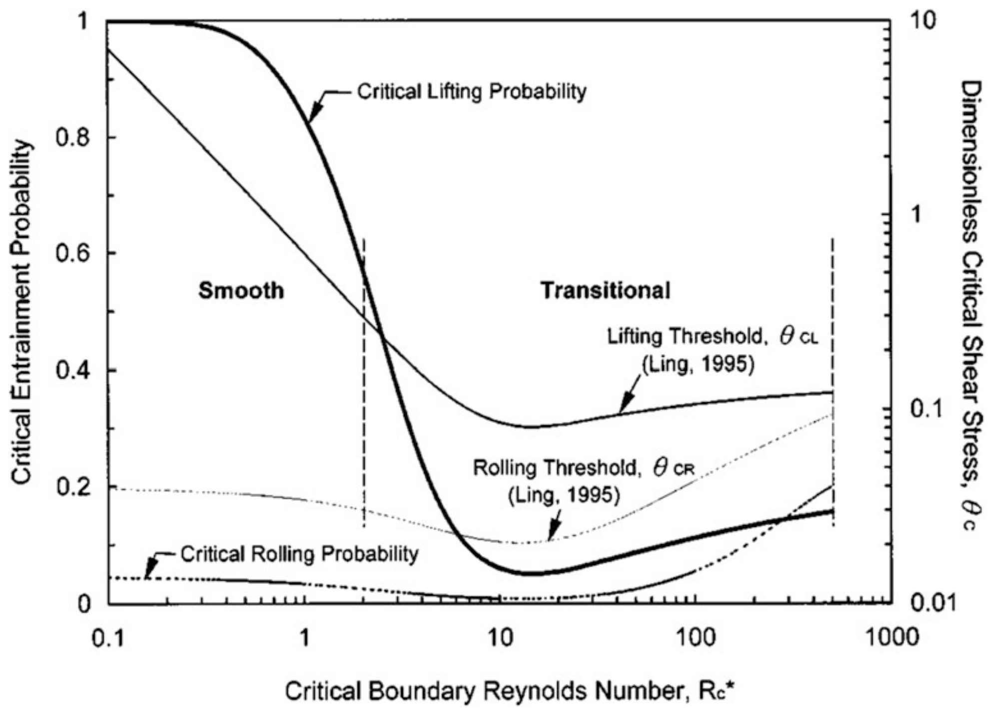
The interaction between fluid and solid particles is greatly influenced by the sediment characteristics. Sediments are commonly divided into cohesive and non-cohesive components. In cohesive sediments, the resistance to erosion depends on the strength of the cohesive bond between the particles. The problem of erosion resistance of cohesive soils is a very complex one, and at present our understanding of the physics of it is still very incomplete. The non-cohesive soils generally consist of larger discrete particles (e.g., sand, pebbles, cobbles, and boulders); the movement of these particles essentially depends on the physical properties of the individual particles, such as their size, shape, and density. The most important physical property of a sediment particle is its size which has a direct effect on its mobility.

There are dozens of sediment transport functions that predict sediment transport based on sediment size, weight, fall velocity, water velocity, channel depth, channel width, slope, roughness, and water temperature. However, in many of these empirical solutions, some assumptions are made. Moreover, in sediment transport, two important concepts are settling rate and boundary shear stress. Settling rate describes the tendency for sediment particles to fall out of suspension. Boundary layer shear stress describes the tendency for a moving fluid to bring sediment particles into suspension.

Sediment entrainment or the threshold for sediment motion

Sediment entrainment is defined as the transition from repose to displacement. The incipient motion of sediment occurs when the stability of a particle is disturbed. Such instability can be attributed to the imbalance of forces (or force moments) caused by the forces exerted on the particle in the flow. The threshold of particle motion is attained for a given ratio between driving and stabilizing forces. Simplifying, the driving forces on a sediment particle resting on other particles on an originally plane horizontal bed are the tractive stress T $\sim$ (horizontal) and the lift force (caused by the Bernoulli effect). The horizontal drag, created by the flow, consists of skin friction acting on the surface of the grain and from drag due to a pressure difference on the upstream and downstream sides of the grain because of flow separation.

The motion of a fluid flowing across its bed tends to move the bed material downstream. A submerged grain on the surface is subjected to a weight force and the hydrodynamic forces. Below some critical hydraulic conditions, the hydrodynamic forces will be so small that particles submerged weight will move very rarely or not at all. However, a slight increase in flow velocity above this hydraulic critical condition will initiate appreciable motion by some of the particles on the bed. This hydraulic critical condition is termed the condition of initiation of motion and is computed in terms of either mean flow velocity in the vertical or the critical bed shear stress (also known as the tractive force or the drag force).



Sediment Transport, Figure 1 Critical entrainment probability and its relationship with shear stress and Reynolds number.

The entrainment of sediment has been investigated by many approaches. One approach is to determine the critical shear stress for incipient motion of sediment. The work of Shields (1936) is the most well-known entrainment criterion. Quantification of the threshold shear stress is the basis for prediction of transport rate in many bed-load equations (e.g., Meyer-Peter and Müller, 1948; Parker, 1979). On the other hand, some researchers support the existence of a range of threshold values for initial sediment movement and thus employ the probabilistic model as an alternative approach to sediment entrainment (e.g., Einstein, 1942; Grass, 1970; Gessler, 1971) and the prediction of bed transport (e.g., Einstein, 1950; Paintal, 1971). The field and laboratory observations also confirm the variability of critical shear stress that can be attributed to a number of random factors (e.g., temporal fluctuations of turbulent flow, heterogeneity of grain size, shape and density, bed-grain geometry, sediment availability, exposure and sheltering effect, bed roughness, etc.).

Bed-load transport

When the bed shear stress exceeds a critical value, sediments are transported in the form of bed load and suspended load. The sediment transport rate may be measured by weight, mass, or volume. In practice, the sediment transport rate is often expressed per unit width and is measured either by mass or by volume. To make predictions about the conditions under which sediment will be transported, it is common to use Shields stress and the particle Reynolds number (Figure 1). The Shields stress conciliates settling rate and boundary layer shear stress in order to predict when a moving fluid will transport sediment. The Reynolds number (Re) predicts the extent of turbulence in a fluid based on flow velocity (u), characteristic length (l) which represents flow geometry, fluid density (r), and fluid viscosity (μ). Turbulent flow has Re $> 2{,}000$, and laminar flow has Re < 500. Flow with Re between 500 and 2,000 is transitional.

In dimensionless terms, the condition for bed-load motion is when bed shear stress (τ_0) exceeds a critical value (τ_0)c:

$$\tau_0 = (\tau_0)c$$

The Shields parameter is the nondimensional number used to calculate the initiation of motion of sediment in a fluid flow:

$$\tau_* = \frac{\tau_0}{(\rho_s - \rho)gd_s}$$

where τ_* is dimensional shear stress, ρ_s is the density of sediment, ρ is the density of fluid, g is acceleration due to gravity, and d_s is a characteristic particle diameter of the sediment.

Table 1 presents a summary of empirical and semiempirical correlations of bed-load transport.

Yallin (1963, 1972) developed a bed-load equation incorporating reasoning similar to Einstein (1942, 1950), but with a number of refinements and additions.

Sediment Transport, Table 1 Empirical and semiempirical correlations of bed-load transport (Chanson, 1999)

Reference (1)	Formulation (2)	Range (3)	Remarks (4)
Boys (1879)	$q_s = \lambda\tau_o(\tau_o - (\tau_o)_c)$		λ was called the characteristic sediment coefficient
	$\lambda = \frac{0.54}{(\rho_s - \rho)g}$ Schoklitsch (1914)		Laboratory experiments with uniform grains of various kinds of sand and porcelain
	$\lambda \propto d_s^{-3/4}$ Straub (1935)	$0.125 < d_s < 4$ mm	Based upon laboratory data
Schoklitsch (1930)	$q_s = \lambda'(\sin\theta)^k(q - q_c)$ $q_c = 1.944 \times 10^{-2} d_s(\sin\theta)^{-4/3}$	$0.305 < d_s < 7.02$ mm	Based upon laboratory experiments
Shields (1936)	$\frac{q_s}{q} = 10\frac{\sin\theta}{s}\frac{\tau_o - (\tau_o)_c}{\rho g(\mathbf{s}-1)d_s}$	$1.06 < s < 4.25$ $1.56 < d_s < 2.47$ mm	
Einstein (1942)	$\frac{q_s}{\sqrt{(s-1)gd_s^3}} = 2.15\exp\left(-0.391\frac{\rho(\mathbf{s}-1)gd_s}{\tau_o}\right)$	$\frac{q_s}{\sqrt{(s-1)gd_s^3}} < 0.4$ $1.25 < s < 4.25$ $0.315 < d_s < 28.6$ mm	Laboratory experiments. Weak sediment transport formula for sand mixtures. Note: $d_s \approx d_{35}$ to d_{45}
Meyer-Peter (1951), Meyer-Peter and Müller (1948)	$\frac{\dot{m}^{2/3}\sin\theta}{d_s} - 9.57(\rho g(s-1))^{10/9} = 0.462(s-1)\frac{(\rho g(\dot{m}_s)^2)^{2/3}}{d_s}$	$1.25 < s < 4.2$	Laboratory experiments. Uniform grain size distribution
	$\frac{q_s}{\sqrt{(s-1)gd_s^3}} = \left(\frac{4\tau_o}{\rho(s-1)gd_s} - 0.188\right)^{3/2}$		Laboratory experiments. Particle mixtures. Note: $d_s \approx d_{50}$
Einstein (1950)	Design chart $\frac{q_s}{\sqrt{(s-1)gd_s^3}} = f\left(\frac{\rho(s-1)gd_s}{\tau_o}\right)$	$\frac{q_s}{\sqrt{(s-1)gd_s^3}} < 10$ $1.25 < s < 4.25$ $0.315 < d_s < 28.6$ mm	Laboratory experiments. For sand mixtures. Note: $d_s \approx d_{35}$ to d_{45}
Schoklitsch (1950)	$\dot{m}_s = 2500(\sin\theta)^{3/2}(q - q_c)$ $q_c = 0.26(s-1)^{5/3}d_{40}^{3/2}(\sin\theta)^{-7/6}$		Based upon laboratory experiments and field measurement (Danube and Aare rivers)
Nielsen (1992)	$\frac{q_s}{\sqrt{(s-1)gd_s^3}} = \left(\frac{12\tau_0}{\rho(s-1)gd_s} - 0.05\right)\sqrt{\frac{\tau_o}{\rho(s-1)gd_s}}$	$1.25 < s < 4.22$ $0.69 < d_s < 28.7$ mm	Re-analysis of laboratory data

Note: $\dot{m}$ = mass water flow rate per unit width; $\dot{m}_s$ = mass sediment flow rate per unit width; q = volumetric water discharge; q_s = volumetric sediment discharge per unit width; $(\tau_o)_c$ = critical bed shear stress for initiation of bed load

Yang (1972, 1973) approached the total transport from the energy expenditure point of view and related the transport rate to stream power. Shen and Hung (1971) derived a regression equation based on laboratory data for the sand-sized particles. Using the same concept, Ackers and White (1973) defined sediment transport functions in terms of three dimensionless groups, namely, size, mobility, and transport rate of sediments. Their functions are based on flume data carried out with flow depths up to 0.4 m. One of the most extensive field and laboratory studies of sediment transport is that by Van Rijn (1984). He has presented a method which enables the computation of the bed-load transport as the product of the saltation height, the particle velocity, and the bed-load concentration.

The theoretical equation for the distribution of suspended sediment in turbulent flow has been given by H. Rouse. Further useful information on the modification of the theory can be found in Einstein and Chien (1955), Vanoni (1984), Hassanzadeh (1985, 1979), and many others (Graf, 1971; Graf and Altinakar, 1998; Raudkivi, 1976). Mei et al. (1994) also reported a study on the hyperconcentrated fluid mud in rivers.

Several researchers have proposed formulas to estimate the characteristics of the bed-load layer (Table 2).

Settling velocity

The settling velocity of sediment is one of the key variables in the study of sediment transport. It hinges on the type of flow (laminar, transitional, and turbulent) of the fluid that transports the particle. Stokes' settling theory describes the velocity of a spherical particle settling through a fluid – depending on a balance between the drag force and the gravitational force. At the settling velocity,

Sediment Transport, Table 2 Bed-load transport rate calculations (Chanson, 1999)

Reference (1)	Bed-load layer characteristics (2)	Remarks (3)
Fernandez-Luque and van Beek (1976)	$\frac{V_s}{V_*} = 9.2\left(1 - 0.7\sqrt{\frac{(\tau_*)_c}{\tau_*}}\right)$	Laboratorydata $1.34 \leq s \leq 4.58$ $0.9 \leq d_s \leq 3.3\,\text{mm}$ $0.08 \leq d \leq 0.12\,\text{m}$
Nielsen (1992)	$C_s = 0.65$ $\frac{\delta_s}{d_s} = 2.5\left(\tau_* - (\tau_*)_c\right)$ $\frac{V_s}{V_*} = 4.8$	Simplified model
Van Rijn (1984a, 1993)	$C_s = \frac{0.117}{d_s}\left(\frac{v^2}{(s-1)g}\right)^{1/3}\left(\frac{\tau_*}{(\tau_*)_c} - 1\right)$ $\frac{\delta_s}{d_s} = 0.3\left(d_s\left(\frac{(s-1)g}{v^2}\right)^{1/3}\right)^{0.7}\sqrt{\frac{\tau_*}{(\tau_*)_c} - 1}$ $\frac{V_s}{V_*} = 9 + 2.6\log_{10}\left(d_s\left(\frac{(s-1)g}{v^2}\right)^{1/3}\right) - 8\sqrt{\frac{(\tau_*)_c}{\tau_*}}$	For $\frac{\tau_*}{(\tau_*)_c} < 2$ and $d_s = d_{50}$ Based on laboratory data $0.2 \leq d_s \leq 2\,\text{mm}$ $d > 0.1\,\text{m}$ $Fr < 0.9$
	$C_s = \frac{0.117}{d_s}\left(\frac{v^2}{(s-1)g}\right)^{1/3}\left(\frac{\tau_*}{(\tau_*)_c} - 1\right)$ $\frac{\delta_s}{d_s} = 0.3\left(d_s\left(\frac{(s-1)g}{v^2}\right)^{1/3}\right)^{0.7}\sqrt{\frac{\tau_*}{(\tau_*)_c} - 1}$ $\frac{V_s}{V_*} = 7$	$d_s = d_{50}$. Based on laboratory data $0.2 \leq d_s \leq 2\,\text{mm}$ $d > 0.1\,\text{m}$ $Fr < 0.9$

Notes: V_* = shear velocity; $(\tau_*)_c$ = critical Shields parameter for initiation of bed load

the drag force (F_d) on the sphere is balanced by the excess of the gravitational force (F_g):

$$F_d = \left(Cd\rho_f A_p w_s^2\right)/2$$

$$F_g = \left(\rho_p - \rho_f\right) w_s g$$

Introducing R, the submerged specific gravity:

$$R = \frac{\rho_p - \rho_f}{\rho_p}$$

Introducing the drag coefficient defined in terms of the Reynolds number (Re):

$$\text{Re} = \frac{w_s D}{v}$$

Producing an equation for the (Stokes) settling velocity (w_s):

$$w_s = \frac{RgD^2}{18v}$$

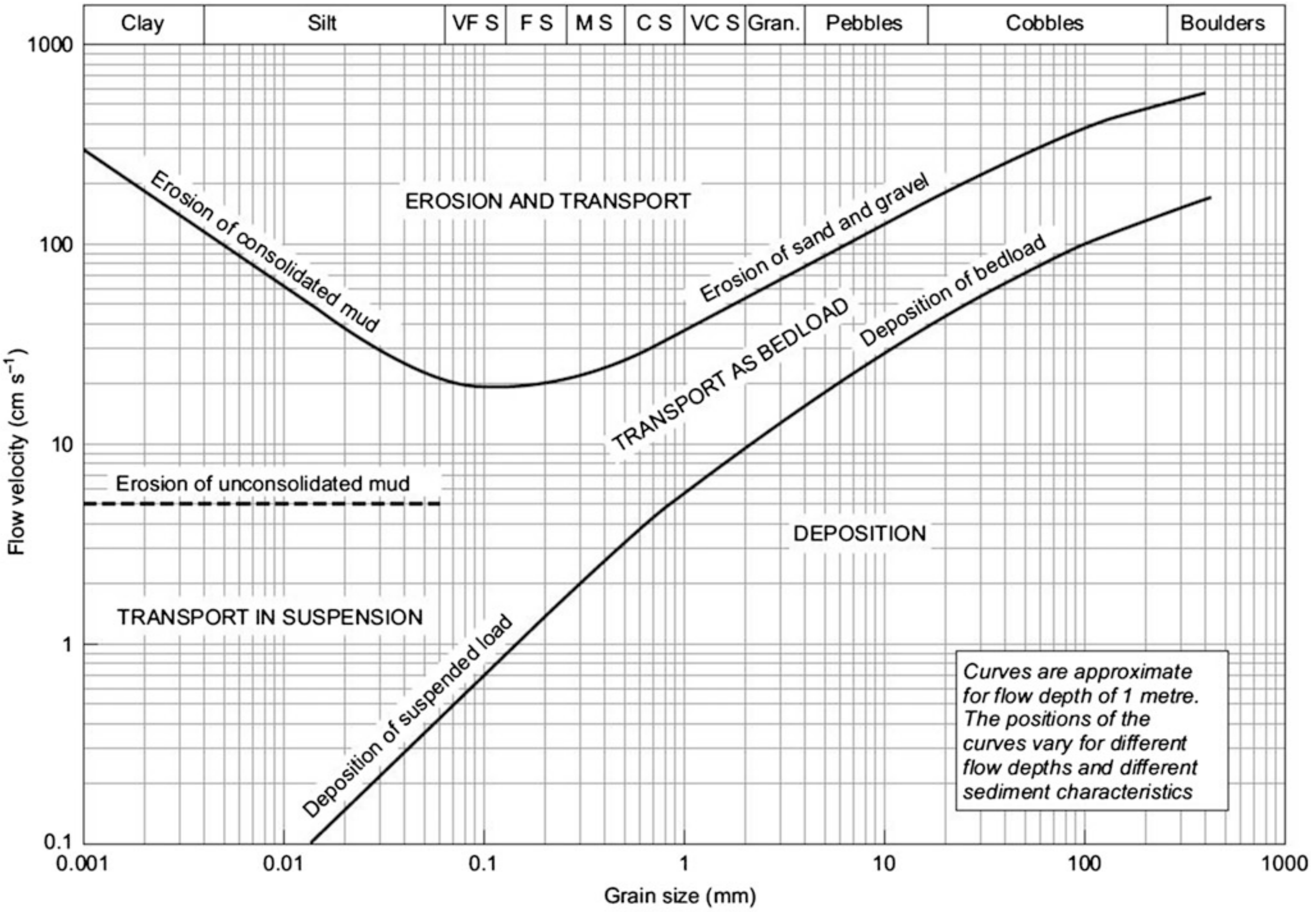
where Cd is the drag coefficient, ρ_f is the density of the fluid, ρ_p is the density of the particle, A_p is the area of the particle, w_s is the particle velocity, g is the gravitational acceleration, R is the radius of the spherical object, and v is kinematic viscosity.

The fall or settling velocity of a particle is assumed to be a steady-state motion. It is also a function of size, shape, density, and viscosity of fluid. In addition, it depends on the extent of the fluid in which it falls, on the number of particles falling, and on the level of turbulence intensity. Often the estimation of settling velocity of sediment has been done by applying predictive formulas developed by assuming the grains to be spheres. It is well known that the shape of natural sediment particles departs from a sphere. This departure will have some consequences, one being that the settling velocity will be lower than that of a sphere with the nominal diameter. Due to the practical implications of this difference, several formulas have been proposed to calculate the settling velocity of natural nonspherical grains (e.g., Graf, 1971; Zanke, 1977; Hallermeier, 1981; Dietrich, 1982; van Rijn, 1984; Swamee and Ojha, 1991; Julien, 1995; Cheng, 1997; Soulsby, 1997; Ahrens, 2000). Also, it is the empirical work of Bagnold (1941) that particularly focuses on aeolian sediment transport.

The deviation of a particle's shape from a sphere is generally quantified by a shape factor. The most commonly used is Corey's shape factor, which is given by

$$csf = \frac{c}{\sqrt{ab}}$$

where a, b, and c are the longest, intermediate, and shortest axes of the particle.



Sediment Transport, Figure 2 The Hjulstrom diagram shows the relationship between the velocity of water flow and the transport of loose grains. Once a grain has settled, it requires more energy to start it moving than a grain that is already in motion. The cohesive properties of clay particles mean that fine-grained sediments require relatively high velocities to re-erode them once they are deposited, especially once they are compacted.

The flows that are required to pick up grains of certain sizes have been extensively studied empirically, and the results are plotted in Hjulstrom diagrams (Figure 2). A Hjulstrom diagram is an empirical measure of the minimum velocity required for moving particles of different sizes. The diagram shows grain entrainment on a plot of log grain size versus log flow speed. Note that larger grains require higher flows, in general. The water speed that is required to transport a grain is called the critical velocity.

Furthermore, the equation of Ferguson and Church (2004) also expresses settling velocity (w in ms^{-1}) as a function of sediment size D in m:

$$w = \frac{RgD^2}{C1 + \sqrt{0.75\ C2\ Rg\ D^3}}$$

where R = submerged specific gravity (1.65 for quartz in water), g = acceleration due to gravity (9.8 m s^{-2}), n = kinematic viscosity of the fluid (1.0×10^{-6} kg m^{-1} s^{-1} for water at 20 °C), and C1 and C2 are constants. For natural sand grains, Ferguson and Church (2004) recommend C1 = 18 and C2 = 1.

Sedimentary structures

Sedimentary structures directly linked with sediment transport are parallel bedding, ripples, dunes, sand waves, and graded bedding. Structures form on the surface of a bed when topography influences the strength of the flow (and thus the strength of the Bernoulli effect). Erosion occurs where flow is strongest and directed into the bed. Deposition occurs where flow is slower. Deposition ordinarily creates laminae that are parallel to the depositional surface. Small ripples have small laminae that dip downstream because that is where deposition occurs. Flat beds have flat laminae. Large dunes have coarser laminae that dip downstream.

Summary

The purpose of this work is to briefly summarize the complex processes, mechanisms, and physics involved in sediment transport. This contribution provides a short review of sediment entrainment features, empirical and semiempirical correlations of bed-load transport, and formulas used to calculate settling velocity. In addition, it provides examples of sedimentary structures that are the direct stratigraphic translation of sediment transport.

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Cross-references

Sediment Grain Size
Sediment Resuspension

SEDIMENTARY STRUCTURES

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Synonyms

Bedding; Bedforms

Definition

Sedimentary structures. Primary or mechanical structures formed by physical processes in the sedimentary environment.

Bedforms. Primary sedimentary structures formed by the interactions between turbulence of flow and sediment grains.

Antidunes. Large-scale structures looking like plane beds formed after destruction of bedforms in the upper part of the higher flow regime.

Megaripples. Large-scale sedimentary structures formed in the upper part of the lower flow regime.

Ripples. Small-scale sedimentary structures formed in the lower part of the lower flow regime.

Bedding planes. Surface sedimentary structures.

Bedding. Layering characteristics of the sequence.

Introduction

Sedimentary structures are surficial or internal, megascopic, three-dimensional features of sediments or sedimentary rocks (Pettijohn and Potter, 1964). These structures have been called mechanical or primary structures (Potter and Pettijohn, 1977) due to their formation by physical processes. In the modern environment, flow regimes at varying speeds and velocities produce different sedimentary structures that are called bedforms. Sands are deposited in a diverse suite of ripples, megaripples, sand waves, rill marks, rhomboid marks, backwash ripples, swash marks, and current crescents in the central basin and marginal areas of an estuary (Elliot, 1983). Ripple bedforms have a tendency to occur in the intertidal areas of mid-channel bars or point bars where tidal current velocities are at a minimum, whereas megaripples and sand waves are confined to the depressed zones of tidal sand bars.

Small ripples are formed by an increase in flow velocity, and these migrate in the direction of flow. With a continuous increase in flow velocity, the small ripples may enlarge and change slope giving rise to megaripples. Plane beds and antidunes gradually form as a result of the destruction of megaripples at higher flow rates. Ripples and megaripples are the most commonly observed bedforms in the estuarine environment (Davis, 1983) (Figures 1, 2, and 3).

Sedimentary Structures, Figure 1 Lingoid Ripples.

Sedimentary Structures, Figure 2 Rill Marks.

Bedforms characters

The sand-dominated middle to lower stretch of the estuary is significantly important where intertidal sand bodies with numerous bedforms of different scales are exposed after each tide. The exposed portion of the estuarine bed and the intertidal mid-channel bars exhibit various bedforms of tidal origin. Inherent unsteadiness in flow conditions, reversals of tidal currents, and bedform-flow interactions are the causal factors for the frequent changes in bedform architecture (Middleton, 1965; Conybeare and Crook, 1968). Thus, bedforms, though not static, are permanent features which often display a quasi-equilibrium form under effects of variable tidal dynamics. Smaller bedforms quickly change their orientations, but large

Sedimentary Structures, Figure 3 Wave Ripples.

sand waves do not, in response to changing flood-ebb conditions (cf. Dalrymple et al., 1978; Collinson and Thomson, 2006).

Types of bedforms

Three different scales of bedforms (i.e., small scale, intermediate scale, and large scale) are recognizable by different geomorphic units in mesotidal and macrotidal estuaries. The small-scale bedforms include all varieties of ripple marks. The intermediate-scale bedforms include two different types of megaripples, and the large-scale bedforms include sand waves.

Morphology of the bedforms

Large-scale bedforms

Sand waves

These features are the largest-scale bedforms observed on the mid-channel bar surface. They are flood-oriented, two-dimensional forms and appear mainly on the floor of the flood-dominated portions of the bar surface. They lack scour pits in their troughs and spurs on the slip faces. Ripples superimposed on the sand waves have been partially planed off by the preceding ebb flow.

The intermediate-scale bedforms

Straight-crested megaripples

The mid-channel bar surface is extensively sculptured by trains of straight-crested megaripples. They are parallel to each other over considerable distances but exhibit minor sinuosity. These megaripples have broadly convex-up profiles, and their crestal heights remain more or less constant without showing any well-developed scour pits in the troughs. The troughs contain small ripples with crestal orientations perpendicular to the megaripple crests. This is because of the generation of secondary flow along the troughs of megaripples during falling water levels (Das and Bhattacharya, 2000).

Undulatory megaripples

Undulatory megaripples possess long wavy or undulating crests and are devoid of well-developed scour pits in front of their slip faces. In this respect, they differ from lunate megaripples of Reineck and Singh (1980) or type II megaripples of Dalrymple et al. (1978) in which the crest line is broken and megaripples possess distinct scour pits in front. Both in-phase and out-of-phase arrangements of undulations are present in a single megaripples train. The crests have forward tonguelike projections and steep profiles in contrast to those of the straight-crested megaripples. The megaripples surface is ornamented by linguoid ripples, which are produced by emergence. Allen’s (1968) category of megaripples corresponds to this type of bedform.

Small-scale bedforms

Ripples

These small-scale bedforms occur under the direct influence of both ebb and flood-tidal currents as well as under the influence of any one of these currents where they originate along troughs of megaripples. The chief controlling factors for their formation are the general slope of the troughs, strength of tidal currents, and direction of wings. The small-scale ripples are of various types as described below.

Straight-crested ripples

These ripples have more or less straight crests and crestal trains of successive ripples which run parallel for a few meters. The surface undulation is very gentle because of low ripple heights.

Linguoid ripples

These ripples exhibit a crescentic pattern in plan. The crests are arcuate to tonguelike and have forward closures. These ripples produce appreciable surface undulations. The lee slopes can be easily measured in the field.

Wave ripples

These ripples are asymmetrical in plan and closely associated with straight-crested ripples. They typically have steep lee slopes and very gentle stoss slopes. In addition, they exhibit a positive linear correlation between their length and height measurements.

Flat-topped ripples

These bedforms occur as superimposed ripples over larger bedforms and along troughs of megaripples.

The flattening of the crests is evident for both linguoid and asymmetric wave ripples. The troughs are very narrow compared to the width of the crests. Because of the greater flattening of the ripple crests, the crestal lines often get obliterated; the crestal width increases with a corresponding decrease of trough width. The mechanism of formation of flat-topped ripples is attributed to scouring by tidal currents.

Double-crested ripples

These ripples generally contain double crestal trains with identical spacing of 5–6 mm. The ripples are typically asymmetric with almost straight crests which often terminate laterally against the linguoid ripples. These form as a result of changes in water depth with changing tidal level. The double-crested ripples are often supposed to be diagnostic of intertidal flats (Reineck and Singh, 1980; Terwindt, 1988). Klein (1970) explained the mechanism of formation of secondary currents over the primary ones from estuarine environments.

Ladder-back ripples

These are interference ripples in which two sets of ripples maintain an oblique to perpendicular relationship with their crestal trains. Reduction of water depth particularly during the ebb phase controls the size and orientation of the current ripples. With the decline of water level, the size of the current ripples decreases resulting in superimposition of the small ripples over the larger sets. Many different configurations exist for the ladder-back type to complex network of ripple trains.

Backwash ripples

The backwash ripples occupy the highest topographic areas and concentrate in the regions of maximum advance of wave swash in the swash platform of the estuarine environments. These are gentle undulations parallel to river banks and formed away from the river channel margin. The bedform is prominent due to variations of color from the crests to the troughs of the ripples. They are sinuous in plan and are generally asymmetrical in profile with lee slope direction toward the river channel. Dark minerals, mostly biotite, concentrate along the troughs of the ripples, whereas light-colored quartzo-feldspathic minerals mark the ripple crests. Thus, instead of being marked by their relief, these ripples are characterized by sinuous, alternate light and dark color bands.

Swash marks

These are tiny, curved ridges or markings on a sandy swash platform. The curved ridges with their convexity landward mark the maximum advance of wave swash. The ridges are generally of insignificant heights and exhibit strike-wise continuation for several meters although with minor breaks at places. The ridges are generally comprised of very fine sand grains. Swash marks result from the lobate fronts of dying waves during backwash and mark the line of maximum advance of wave swash. In the estuarine areas, the swash marks document the limit of the outer bank of the estuaries, and as a result their alignment is at right angles to the shoreline alignment. Hence, the swash marks in an estuary occur almost at right angles to their counterparts on the sea beach.

Rill marks

Rill marks are dendritic erosional structures on a sandy swash platform made by a system of small rivulets originating from the flow of a thin layer of water during a falling water stage. Rill marks are of various forms and dimensions, and their morphological variations are primarily controlled by local topography, slope of sediment surface, grain size, and water flow. Rill marks are quite abundant in sandy platforms of estuarine environments.

Partially conical rill marks

These small bedforms appear in the form of partially developed conical depressions whose walls are sculptured by fine rills. The cones are about 15 cm across. Water drained from the conical rills unites to form larger rill marks (70–80 cm long) with accumulation tongues downslope. A sudden change in slope of the platform is marked by a change in the morphological variety of rill marks.

Bifurcating rill marks

These rill marks often exhibit downslope bifurcation and a sinuous or meandering pattern. The bifurcation is often quite overt with the last bifurcations opening in the downslope and downcurrent direction. These rill marks are confined to a slope angle ranging from 2° to 3° and extend for a distance of 3–8 m on sandy platforms.

Branching rill marks

Branching rill marks are composed of small rill systems bundled together to form a broad channel. They have very prominent bifurcations that yield a dendritic pattern. The finer rills unite together downslope and are often confined to an eroded broad channel whose walls stand 3–4 cm high from the rill floor, which is also characterized by coarser lag materials of fragmental shells and mud pellets. Branching rills occur on a slope angle ranging from 3° to 6°.

Rhomboid marks

Rhomboid marks are formed on swash platforms in estuarine regions. They are diamond-shaped structures with their long diagonals aligned at right angles to the longitudinal profile of a river. There are two different sets of rhomboid marks which appear as superimposed large-scale and small-scale reticulate patterns on the sandy surface. The smaller set ranges from 2 to 3 cm along their longer diagonals and 0.8–1.2 cm along their shorter diagonals. The larger set has longer diagonals about 1 m and shorter diagonals about 45 cm. Rhomboid marks have positive relief of a few mm to less than 1 cm from the normal sediment surface. Rhomboid marks originate from a relatively high-velocity condition of the wave backwash when the depth of the water sheet is less than 1.5 cm. They appear on the sandy platform when the wave backwash moves down leaving the area exposed.

Current crescent

These are crescent-shaped (U or V shaped) structures which widen in the flow direction. They have been shown to form around dead gastropod or pelecypod shells with their arms open downslope toward the river direction (Bhattacharya, 1993). The tapering end of the dead gastropod shells always points downcurrent and in a downslope direction. Current crescents form around scattered tools of permeable or impermeable bluff bodies like pellets, shell fragments, mineral matters, lithic fragments, vegetation hummock, and decomposable organic matter in the aqueous estuarine environment.

Internal sedimentary structures

These structures in the estuarine environment are only seen in several trench sections cut through the mid-channel bar and washover flat sediments during short periods of their surface exposure at falling water stages. Internal sections from dugout trenches of mid-channel bars exhibit large-scale tabular cross-stratifications with dip directions controlled by the dominating flow. Some trench sections in the mid-channel bars clearly reveal penecontemporaneous deformations of sandy laminae toward the crestal part of the arrested megaripples. The deformations are marked by slightly disturbed layers with minor undulations or folding. The megaripples bedding is followed upward by parallel laminations as internal structures with distinct tidal bundles characterized by alternations of sand-mud couplets and sets of horizontal stratifications. In some occasions, convex-upward reactivation surfaces are present within cross-bedded units.

Sedimentary units depicting a number of internal sedimentary structures have been recognized in washover flats following sequences from bottom to top.

Hummocky cross-stratification

These are low-angle (2–3°) undulating cross-stratifications. The lamina sets (15–20 cm thick) are both concave and convex upward with wavelength of 1–2 m and height of 6–15 cm. Texturally, the hummocky cross-stratifications are comprised of sand-sized particles.

Planar tabular cross-stratification

The planar tabular cross-beds form an isolated set in between the underlying hummocky cross-stratifications and the overlying sand-mud laminated units. The foreset dips landward at an angle of 15–18° and therefore indicates their flood-tidal origin. The mud drapes and mud laminae punctuated within foreset laminae refer to slackening structure. The planar cross-beds are not laterally persistent. They grade to flaser beds and farther away (15–20 m) into parallel-laminated units.

Flaser bedding

These structures are cross-stratified ripple bedding containing thin streaks of mud in the crests and troughs of the ripples. Most flaser laminations are the wavy flaser type of Reineck and Wunderlich (1968) in which the mud flasers are concave upward when they occupy the crests of ripples and concave downward in the overlying ripple crests. The flaser-bedded unit continues laterally for a length of 4–5 m. The mud flasers are often of several millimeters in thickness.

Conclusion

Sedimentary structures in estuarine environments have been utilized for interpreting the hydrodynamic conditions of their formation. These structures reflect the effects of macrotidal regimes, moderate wave energy, and longshore currents (at river mouth bars). The various scales of tidal cycles involving neap-spring and ebb-flood, together with wave swash and backwash, impart some modifications in the bedforms. Internal manifestation of bedforms is the direct consequence of surface features, and many of these are interpreted as features of the tidal domain. The morphology of the bedforms of Hugli-Matla estuary in India, with some exception, closely resembles those described for the Bay of Fundy, Canada, and Loughor estuary, UK.

It is concluded that the perpendicular-to-shoreline alignment of bedforms like backwash ripples, swash marks, rhomboid marks, current crescents, and internal sedimentary structures of estuarine environments may be interpreted incorrectly as beach features in the rock record. Such a misinterpretation may lead to a 90° error in the mapping of a local palaeo-shoreline at the mouth of an estuary.

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SEICHE

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Synonyms

Eagre; Sea swell

Definition

A seiche is a stationary wave oscillation that causes water surface oscillations in any period and height under the effects of different forces such as an earthquake, wind, barometric pressure, and tide. It was initially described by the Swiss hydrologist François-Alphonse Forel in 1890, who made the first scientific observations in Lake Geneva, Switzerland (Darwin, 1898).

Description

The period of a seiche varies from a few minutes to an hour or more. Wind is the most common seiche producing force, causing water surface heaves against the downwind shore. Windbreaks cause water surface oscillation. The period of a seiche is calculated using Merian's formula:

$$T = \frac{1}{n}\frac{2L}{\sqrt{gd}} \tag{1}$$

where T is the period of the seiche, n is the number of the nodes of seiche, L is the mean length of the basin, d is the mean depth, and g is the acceleration of gravity (Chow, 1964).

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SHANNON-WEAVER DIVERSITY INDEX

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Synonyms

Shannon index; Shannon information index; Shannon-Wiener index

Definition

The Shannon-Weaver diversity index is based on communication theory. The uncertainty is measured by the Shannon Function "H'." This term is the measure corresponding to the entropy concept defined by

$$H' = -\sum_{n=1}^{n} (pi^{*} \ln p_i) \tag{1}$$

where H' is the diversity index, p_i is the proportion of each species in the sample, and ln p_i is the natural logarithm of this proportion (Shannon and Weaver, 1949; Spellerberg and Fedor, 2003; Magurran, 2004).

Description

The Shannon-Weaver diversity index is one widely used index for comparing diversity between various habitats (Clarke and Warwick, 2001). It assumes that individuals are randomly sampled from an independent large population, and all the species are represented in the sample (Shannon and Weaver, 1949). The value of the Shannon-Weaver diversity index usually ranges from 1.5 to 3.5 and only rarely exceeds 4.5.

The principal objective of a diversity index is to obtain a quantitative estimate of biological variability that can be used to compare biological entities in space or in time. This index takes into account two different aspects that contribute to the concept of diversity in a community: species richness and evenness.

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Cross-references

Species Richness

SHELL BEDS

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Synonyms

Coquinas; Lumachelles; Shell gravels

Definition

Dense deposits of biologic hard parts more than 2 mm in size in estuarine and other environments are generally defined as shell concentrations (Kidwell, 1991). They are also known as coquinas, lumachelles, or shell gravels, as well as the more familiar term "shell beds."

Description

Although the term "shell beds" refers to a particular geometric arrangement of biogenic remains, the great variety of shell beds reflects the diverse descriptive approaches used to classify them. The scheme proposed by Kidwell et al. (1986) is based on different field observations of the shell deposits, such as their biofabric, geometry, taxonomic composition and internal structure, which can be measured in the field by nonspecialists. This procedure allows investigators to obtain a range of ecological, hydrodynamic, and topographic data on the mode of the shell bed formation (Fursich, 1995).

There are different classification schemes for shell beds. In the basic approach proposed by Kidwell et al. (1986), shell beds may be plotted in six areas of a schematic ternary diagram: biologic, sedimentologic, and diagenetic processes are the end-members, whereas three mixed areas reflect combinations of these factors. Comparative analysis may yield data on environmental indicators to characterize, according to prevailing shell bed types, the ideal transect from marginal marine to fully marine depositional settings, even if the same type of shell bed may appear in different environments. In general terms, marginal marine environments (e.g., estuaries) exhibit a diverse assortment of biogenic and sedimentologic concentrations. In the intertidal and supratidal settings, for example, biogenic concentrations include channel-margin oyster bars, mussel clumps, levels of deep-burrowing infaunal bivalves in life position, ray pits, bird's nests, and hermit crab-generated beds, whereas sedimentologic concentrations may be produced by lateral migration of channels and by storm surges (shell pavements and spits). The more refined genetic scheme proposed by Kidwell (1991) on the basis of their depositional histories and stratigraphies organizes shell beds into four broad categories: event, composite, hiatal, and lag concentrations. In this manner, the interpretation yields a stratigraphic signature of the shell-rich sedimentary body to identify the final concentration process and a "taphonomic characterization" of the fossil remains to reconstruct their history before and during the concentration event(s). These four types of shell beds are not discrete categories because they intergrade, and each one may be present in the supra-, inter-, and subtidal environments characterizing the estuarine systems.

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Cross-references

Biogenic Sedimentary Structures
Biogenous Sediment
Sedimentary Structures
Stratigraphy of Estuaries

SHELLFISH PRODUCTION

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Definition

Shellfish production involves evaluation of shellfish resources, often collated from annual surveys, listing the wet weight and monetary value of capture fisheries and aquaculture. In ecological terms, production quantifies biological productivity, the amount of organic matter, or its equivalent in dry matter, carbon, or energy which is accumulated over time.

Introduction

World fisheries and aquaculture production have increased over the past five decades at a rate of 3.2 % per year to 148 million tons, worth $217.5 billion USA dollars, with most used for human consumption (FAO, 2012). Highest consumption has been in Asia, where annual consumption reached 20.7 kg/capita. In China, with an expanding economy and increased domestic income, the per capita consumption has reached 31.9 kg per annum. While the production of fisheries and aquaculture has varied considerably between geographical regions, shellfish production in many areas has remained steady. A declining global marine catch over the last few years and the increased percentage of overexploited species have led to both negative ecological effects and ongoing negative social and economic consequences. For many years, China has been the world's leading shellfish and fish exporter, but some countries, such as Vietnam, have experienced rapid growth. Vietnam is now the fourth largest exporter in the world. The largest importers are the USA and Japan. In many parts of the world, fisheries production is small scale and susceptible to fluctuations. New guidelines are being developed to promote good governance as well as inclusiveness, transparency, gender equality, and respect and involvement of stakeholders.

World fisheries production statistics are divided into wild capture and aquaculture, with further divisions into inland and marine production. Using these divisions, it is often difficult to identify and single out the brackish water production in estuaries. Data for mussels, clams, cockles, and scallops are included in the marine–water aquaculture group, where in 2010 they comprised 75.5 % of the total catch, down from 84.6 % in 1980. After 1980, there was an increase in finfish culture which continued to increase at a rate of seven times that of molluscs. Brackish water aquaculture, however, has increased over the past 20 years, and while only contributing 7.9 % of the world production in terms of quantity, it corresponds to 12.8 % of the total value because of the high value of marine shrimps cultured in brackish ponds. Brackish water crustaceans dominate aquaculture production with bivalves making up only a small percentage (2.1 %) of the total (FAO data in 2010). The aquaculture statistics for 2010 are impressive: 2.7 million tons of white leg shrimp; 4.8 million tons of clams, cockles, and arc shells; 4.5 million tons of oysters; and more than 1.5 million tons of mussels and scallops. The question is – are these production rates sustainable?

Catch and aquaculture trends

Over the last 20 years, changes have occurred in catch statistical trends for marine bivalve species. Clams and cockles once formed more than half of the annual capture catch (Figure 6; FAO, 2012), but more recently they have been equalled by scallops, where the most productive fisheries are in coastal offshore habitats rather than estuaries. In estuaries, the production of mussels and oysters appears to have remained steady or even have declined over the past 10 years; however, there appears to be increased scope for future expansion. There is, however, some uncertainty with the FAO database; not all countries identify catches by species, and aquaculture numbers may not be distinguished from capture fisheries.

Global aquaculture production has increased since 1990, although at a slower rate than was recorded in the 1980s and 1990s, with Asia contributing 89 % of the world aquaculture production in 2010. In Europe, increased aquaculture production has been due to cage culture of salmon, with notable declines in bivalves from 61 % of the total in 1980 to 26.2 % in 2010. In North America, bivalve production appears to be declining, whereas in South America, Brazil, and Peru, there has been strong growth. This increase in bivalve production may be due to the formation of the non-fed aquaculture systems, which avoid the problems associated with feed.

Shellfish species in estuaries

Broadly defined, shellfish are edible invertebrates usually molluscs, crustaceans, or echinoderms. While mussels, oysters, scallops, clams, lobsters, and shrimps are well-known shellfish, less recognized groups include sea urchins, sea cucumbers, abalone, and whelks. All have a hard external covering, a shell or exoskeleton which protects them from extremes of environment and/or predators. Shellfish species that live or are cultured in estuaries are generally euryhaline or salt tolerant, the main groups being mussels, oysters, clams, and shrimps. These are further divided into capture species that are harvested from natural populations and aquaculture species grown in shallow embayments or brackish water ponds. Even with the increasing expansion of aquaculture, wild populations remain important in many parts of the world.

Bivalves are a traditional food for many people. Collecting methods have changed over the years, combining traditional methods with new technology. The collection and aquaculture methods for bivalves are summarized by Gosling (2003); wild shellfish are collected by hand, dredges, and metal baskets, but in deeper water they are collected by diving or in small boats. Mussel and clam production is often higher in estuaries than in the open sea, but this may be a consequence of increased food availability or food quality rather than a direct salinity effect (Seed and Suchanek, 1992). Production from a biological and/or ecological viewpoint requires knowledge of the life history of species; this is because it depends on population regulating mechanisms, including recruitment, growth, and reproduction.

Capture species

Oyster fisheries have a long history; for example, the flat oyster *Ostrea edulis* in Europe and the American oyster *Crassostrea virginica* in the USA are the species which contributed most to the commercial harvest. There have been declines in these species over time (Gosling, 2003), and they now make up only a small percentage of the oyster landing from aquaculture (FAO data). Numerous species of mussels are consumed worldwide; many belong to the genera *Mytilus* and *Perna*. The extent of the wild fishery is uncertain because, even where this is claimed, there may be enhancement by transferring seed from natural habitats onto culture beds for ongrowing. Countries that identify wild mussel catches include the Netherlands, Germany, and Denmark in the Wadden Sea. For clams, about 30 different species are collected worldwide, but two species, the surf clam *Spissula solidissima* and the ocean quahog *Arctica islandica*, make up a large proportion of the clam landings; these are characteristic of offshore benthic marine habitats rather than estuaries. Clam species found in USA estuaries include *Mercenaria mercenaria* and *Mya arenaria*, and in Europe, one of the dominant bivalves is the cockle, *Cerastoderma edule*, found from mid-tide to low-tide level in sandy bottom estuaries. Characterized by irregular recruitment, thought to be a result of weather conditions, successful spatfall occurs approximately once every 6 years. Scallops are a high-value seafood, and natural populations have fluctuated widely in estuarine regions as a result of overharvesting, contaminants, and toxic algal blooms. At present there appears to be low production values for scallops in estuaries, contrasting with the successful production of offshore species.

Aquaculture

Aquaculture depends on a ready supply of seed which can be collected from the field or from brood stock held in a hatchery. While the mussel industry depends mainly on wild spat, clam and oyster aquaculture depends on hatchery-produced seed. There may be problems with diseases in hatcheries, but this may be lessened once the shellfish are in the field. During the grow-out phase trays, ropes, platforms, and rearing nets must be cleaned of fouling organisms. Mussel culture systems include bottom culture, poles, and rafts and suspended culture systems using buoys as undertaken in New Zealand and Chile. In Spain, *Mytilus galloprovincialis* is cultured on wooden rafts in the rias of Galicia, where temperatures range between 10 C and 20 °C and the salinity is about 34 psu (Gosling, 2003). Mussels reach harvestable size of 8–10 cm in about 15 months; growth is slowest in summer probably because of a restricted food supply due to stratification of the water column. Mussels grown in Europe are usually depurated before sale.

Oyster culture has expanded worldwide, with the Pacific oyster *Crassostrea gigas* contributing most to the 4.6 million tons cultured annually. China, Japan, Korea, and France are major producers. Seed supply is normally from natural settlement onto artificial collectors, and ongoing culture methods include bottom, rack, and hanging rope culture. Bottom culture involves sticks, mesh bags, or ground culture, and after a pre-growing period, the spat are scraped off, graded, and put back onto the bottom for periods of up to 2 years (Gosling, 2003). For all types of culture, the growing time to harvest ranges between 2 and 4 years depending on local growth conditions.

Clams total more than three million tons a year, including the Japanese carpet shell or Manila clam *Ruditapes philippinarum* (which has been introduced to France, the UK, and Ireland), hard clam *Mercenaria mercenaria*, and blood cockle *Anadara granosa*. Spat are mostly supplied from hatcheries and once they reach 10 mm in length are seeded into substrate where mesh is laid over the bed to preclude crab and bird predation. The shellfish are usually harvested when the shell length exceeds 20 mm, either by hand or mechanical harvester (Gosling, 2003).

Scallops grown in estuaries provide options for aquaculture in many countries, through a combination of natural seeding, collectors, or hatchery-produced spat and ongrowing on the seafloor on rafts or in nets (Shumway and Parsons, 2006). Scallop production in estuaries over the last 10 years has been declining worldwide with mortalities and population fluctuations, especially in short-lived species. The Atlantic bay scallop *Argopecten irradians* and Yesso scallop *Mizuhopecten yessoensis* are cultured in a number of countries, and there is increased interest in polyculture with seaweeds.

Shrimp and crustacean production

The whiteleg or Pacific white shrimp *Litopenaeus vannamei* is the most productive of the internationally introduced marine crustaceans. Native to Mexico, this shrimp lives in the ocean down to 72 m, but juveniles occur in estuaries where temperatures exceed 20 °C all year. Juveniles are captured in estuaries or cultured in ponds, where they reach 30–35 g after about 7 months. There are issues associated with the expansion of the shrimp industry – the removal of mangroves to construct ponds, pollution of coastal waters by pond effluent, salinization of groundwater, and agricultural land (Roth et al., 2008). Brood stock are sea caught, and one eye is ablated, resulting in repeated maturation and spawning. The hatcheries range from small backyard hatcheries to large environmentally controlled purpose-built buildings. Using natural productivity of the ponds, this species costs less to produce than the more carnivorous *Penaeus monodon* where similar culture techniques were used previously. In 2010 the catch was greater than 2.8 million tons. The main producers in 2004 were China (700,000 t), Thailand (400,000 t), Indonesia (300,000 t), and Vietnam (50,000 t). The freshwater prawn *Macrobrachium* is targeted for ongrowing in some countries such as Bangladesh because prawns grown from wild larvae are considered to be of high quality. Juveniles along the Gulf coast recruit to sea grass and mangrove estuaries where they grow in the rich productive estuarine waters providing a refuge from predation.

Of the total catch from both wild capture and farms, crabs comprise only a small proportion (20 %) of the annual crustacean catch, and although the catch statistics are dominated by the Japanese blue crab, the blue crab *Callinectes sapidus*, the Dungeness crab *Metacarcinus magister*, and the mud crab *Scylla serrate* each contributes 20,000 t annually. Although these landings might appear low, they are important because of their high monetary value compared with bivalves. Chesapeake Bay in the USA once had a thriving industry for hard shell and soft shell blue crabs, but more recently the harvests have fluctuated both here and in other regions. There have been a number of attempts to understand the relationships between production and environmental variables, river discharges, wind, temperature, salinity, rainfall, and hurricane events, but few patterns have emerged (Fogarty and Lipcius, 2007). Modelling studies by Mistiaen et al. (2003), however, have confirmed that productivity would be negatively impacted by poor water quality, suggesting that a decline in the oxygen content of the water to 4 mg l^{-1} could lead to a 48 % decline in the harvest with the same level of fishing effort.

Populations of the Dungeness crab *S. magister*, in the Northeast Pacific, have large amplitudes with a 9–10-year cyclic periodicity. The adults breed offshore, but, like some other decapod crustaceans, rely upon nursery grounds to produce the adult stocks which are captured in other locations. Larvae of the Dungeness crabs enter estuaries where they grow faster than their cohorts, occupying other nearshore habitats (Armstrong et al., 2003). These juveniles, especially those from the large estuary zones close to the ports, are important because mortality is low and abundances are high. There is therefore a high economic value of the estuarine zone in stabilizing coastal landings.

Found in most of the Indo-Pacific, the mud crab *Scylla serrata* is a short-lived species, commonly collected for food and more recently cultured in some countries including the Philippines. Associated with mangroves it survives in low salinities down to 20 psu in estuaries. In subtropical and tropical Australia, *S. magister* is highly sought after, but there are size restrictions and no female crabs are taken from Moreton Bay, allowing females to reproduce. Over the years the commercial catch has increased (113.3 t in 2003), but there is a high recreational catch which exceeds the commercial harvest. It is also suggested that the formation of marine reserves provides the potential for allowing exploited species to recover from the effects of fishing (Pillans et al., 2005). Culture techniques have been tried using pens within mangrove areas in the Philippines and sourced with juveniles from the natural habitat (Trino and Rodriguez, 2002). This study found that mixed sex monoculture in the mangroves was feasible and that production was acceptable at stocking densities in the range 0.5–1.5 crabs m^{-2}. It was concluded that this aquaculture venture is possible without the need to remove mangroves to build aquaculture ponds.

Echinoderm (sea cucumber) production

Following worldwide trends for shellfish production, sea cucumbers, which belong to the class Holothuroidea, are highly prized in Asia, where they are sold as trepan or beche-de-mer and have a variety of food and medicinal uses. They occur in nearshore habitats, including muddy shores close to estuaries, sea grass beds, and rocky and coral reefs. Sea cucumber fishing is important to the livelihoods of coastal communities, especially in developing countries. In Japan for hundreds of years, the people of Oki Island used loose stone piles to encourage the sea cucumbers to aggregate, aestivate, and protect juveniles and young (Choo, 2008). In tropical and subtropical waters in Asia, the fisheries are multispecies, whereas in temperate areas there may be a single species such as the *Apostichopus japonicus*. Indonesia is the largest capture fishery for holothuroids and, together with the Philippines, comprises about 47 % of the world landings, averaging 2,572 t wet weight per annum between 2000 and 2005 (Choo, 2008). It is uncertain what proportion of this catch is collected from estuarine regions and compared with fish, crustaceans, and molluscs. These landings represent only a small percentage (less than 0.25 %) of the total landings. Many boom-and-bust sea cucumber fisheries have occurred since the 1950s as markets rapidly expanded, and regional assessments now suggest that 81 % of the sea cucumber stocks have declined due to overfishing, and the average size reduced by as much as a third. In addition the harvesters have moved from nearshore habitats to offshore habitats and into less valuable species (Anderson et al., 2011). These fisheries are likely to be difficult to manage because of incomplete knowledge about their life cycle, aging, and reproductive cycles. Also, while some countries lack regulations (Indonesia and the Philippines), other populations such as those in Japan and Alaska are well managed (Clark et al., 2009).

In the last 20 years, there have been advances in farming practices for sea cucumbers, especially in China, Ecuador, Indonesia, Japan, Malaysia, and the Philippines. China is the largest producer of sea cucumbers; farming and ranching in the Liaoning and Shandong Provinces, landings have exceeded 10,000 t (dry weight) per annum. Pond culture is the preferred method of farming, sometimes employing unused shrimp ponds containing shelters to protect the animals (Chen, 2004). Sea ranching is a less expensive option for some species, such as *Apostichopus japonicus* the prickly sea cucumber, where the temperature, salinity range (28–31 psu), and sufficient natural food are good. In a sea ranch experiment, Zhang and Liu (1998) report a mean output for this species of 273 kg (dry weight) per hectare, at a density of 12.9 individuals m^{-2}.

Management of shellfish production

For both wild populations and aquaculture, there are management techniques aimed at increasing shellfish production. For oysters on the Atlantic and Gulf coasts of North America, the increased shelling of beds and the removal of gastropod and starfish resulted in a 45-fold increase in the production in Long Island Sound Connecticut in the 1990s (Dumbauld et al., 2009). Management is also required for the oyster industry to minimize the effects of shellfish disease and polluting effects of land runoff and urban development. Other improvements have been achieved through transplants, seeding, silt removal, and reef construction to protect brood stock. Over the past 10 years, there has been increasing interest in developing polyculture systems within controlled estuarine systems. In the Netherlands, mussels have been grown with polychaetes, while in Asia sea cucumbers *Holothuria scabra* have been grown with seaweed and shrimp.

In parts of the world where important shellfish species have declined, management regimes have been controlled since the 1960s by Fisheries Acts introduced by governments or local regulations. Thus, many countries have limits such as total allowable catch (TAC) as for cockles in the UK and permits which limit licenses, restrict fishing days or season length, and return small individuals to the sea. Other mechanisms to protect and/or enhance shellfish production are using restoration techniques to develop wetlands and other habitats or artificially seed juveniles into suitable habitat. In Florida, for example, ready-to-set pedi-veligers of the bay scallop are released into estuaries (Leverone et al., 2010). For successful restoration, not only is there a need for ready supply of viable or healthy spat, but the sites must be in areas where individuals are likely to survive, grow, develop, and maintain sustainable populations. There is therefore a need to better understand the biology of individual target species and how they respond to environmental stressors.

Summary

Estuarine shellfish production is predicted to remain steady or increase over the next 20 years despite widespread belief that global change could fundamentally change the nature of shallow intertidal habitats (Allison et al., 2011). The threats to increased production include increased incidence of pollution events, eutrophication, and toxic algal blooms (Burkholder and Shumway, 2011). There are, however, other threats; for example, invasive or transplanted species such as the Manila clam *Ruditapes philippinarum* or oysters *Crassostrea gigas* may replace native species. While these species can become important dietary components, there is currently little understanding about how such introductions could, in the long term, affect the functioning of estuarine systems. According to Dumbauld et al. (2009), unlike other anthropogenic influences, aquaculture systems do not degrade water quality. Thus, together with whole ecosystem management, the increased use of modern tools, genetics, breeding, improved hatchery techniques, GIS, and modelling, there will be ongoing developments in production techniques which should support millions of people around the world who are employed in the shellfish industry.

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Cross-references

Bivalve Aquaculture
Estuarine Habitat Restoration

SHORE PROTECTION

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Synonyms

Shore erosion control

Definitions

Shore protection along estuarine coasts generally occurs when shorelines are actively eroding. Shoreline erosion, a natural process, is primarily a function of a rising sea level and the impinging local wave climate.

Introduction

Many of the processes that govern erosion (and accretion) on the open ocean coasts also apply to estuarine coasts, but compared to the typically long linear nature of open coasts, estuarine coasts are more sheltered and exhibit a more irregular configuration. Estuarine coasts often display very distinct geomorphic compartments containing a complex mix of resources that may vary from compartment to compartment. The relatively lower wave energy conditions found along estuarine coasts create unique environments that foster habitats and ecological communities, such as marshes and mudflats, not typically found on open coasts (NRC, 2007).

Extent

Just about every estuarine shoreline can experience erosion, but typically when fetch exposures exceed about 1.6 km, land loss by wave action becomes more common (Hardaway and Byrne, 1999). Documented land loss from shoreline erosion is found in numerous estuaries around the USA including Chesapeake Bay, Long Island Sound, Delaware Bay, Puget Sound, and Mobile and Galveston Bays. Land loss in Chesapeake Bay over the 100-year period from 1850 to 1950 is estimated at over 190 km^2 (Singewald and Slaughter, 1949; U.S. Army Corps of Engineers, 1973; Byrne and Anderson, 1978). The response by land owners is to protect their property.

Types

Techniques used to address erosion along estuarine coasts may be placed into broad categories, such as those proposed by Nordstrom (1992), Rogers and Skrabel (2001), and Rogers (2005). Most guidelines and reports on shore protection employ the same basic concepts to discuss approaches such as structural or "hard" methods versus nonstructural or "soft" approaches (USACE, 1981, 1984; New York Sea Grant, 1984; Virginia Marine Resources Commission, 1989; Ward et al., 1989; Pile Buck, 1990; Maryland Department of Natural Resources, 1992; Eurosion, 2004).

Shore protection methods along estuarine coasts are wide ranging but can be grouped into two basic types, defensive and offensive. These are described below.

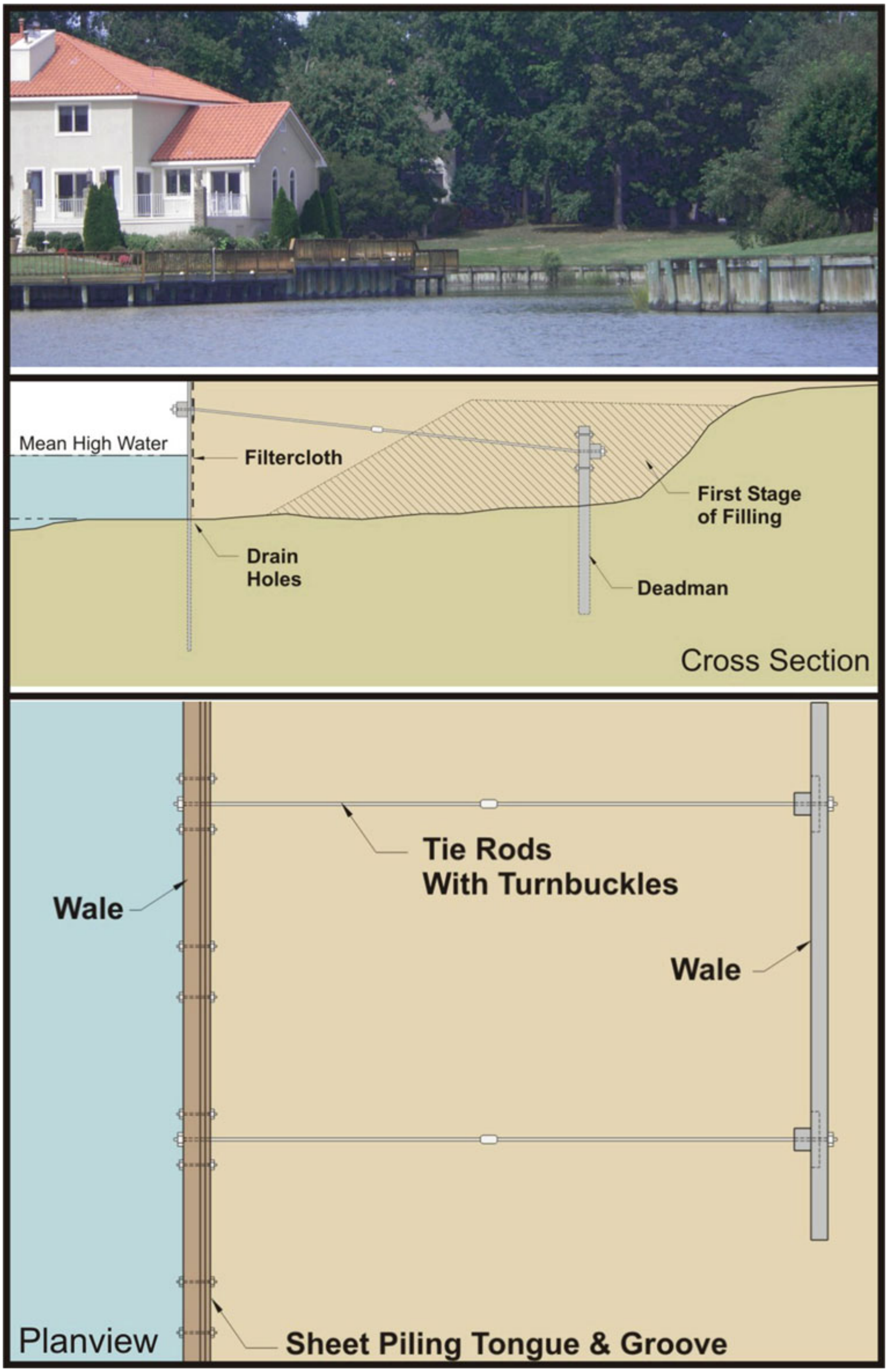
Defensive

Perhaps the most common method of estuarine shoreline protection is to harden the shore with bulkheads, seawalls, or revetments. The primary goal of hardening the shore is to protect the coast from wave attack by creating a barrier to the erosive forces, waves, and currents. According to the NRC (2007), traditional hardening methods often utilize local materials such as stone, wood, and concrete and are built using techniques familiar to local marine contractors and waterfront property owners. While wood bulkheads are common in many areas, stone walls (riprap revetments), constructed of local rock, have become more widely used in areas where rock is available.

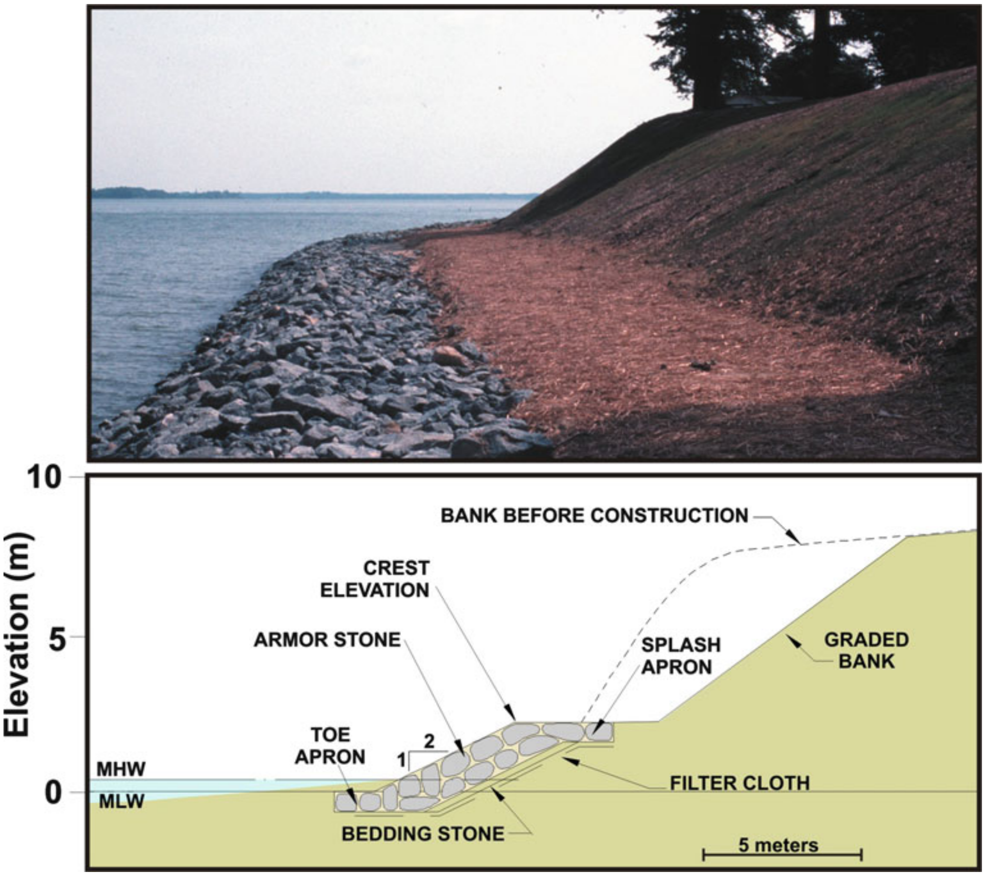
Bulkheads, typically made of wood, are actually retaining walls that consist of the elements shown in Figure 1. The wood is usually treated for longevity in the estuarine environment. The vertical sheet piles are usually tongue and groove and are either driven or "jetted" into place. Horizontal members called wales run along the face of the sheet piling in one or two levels. Tie rods or tie backs are positioned every 1.8–2.4 m and anchored landward to hold the pilings or wales as shown. The tie rods are often placed in conjunction with outside pilings for stability. For a typical bulkhead design, the sheet piles should be in the substrate at least as much as the length above. Also, the structure should allow for groundwater and wave overtopping to be released through the wall via weep holes or other drainage feature.

Some walls are made of concrete and rely on a foundation or footing for stability. They may be referred to as seawalls although this is generally afforded nomenclature for open ocean coasts. The footing is built first, and then the vertical wall is built upon it. These must be properly connected with re-bar or other connecting member. Having them both poured in place at the same time is best. The wall is a gravity structure meant to withstand wave action and hold the land side in place. As with bulkheads some type of drainage system to allow for groundwater and wave overtopping may be required.

Sloped stone walls are commonly called rock revetments, although revetments can be made of other materials such as concrete units, concrete mats, or gabions. Stone revetments armor the front face of the eroding shoreline. They are commonly constructed with one or more layers of graded riprap (Figure 2). The eroding bank is often graded to provide a subgrade for the placement of filter fabric, bedding stone, and then armor. Two layers are preferred, but one made on large stone can be used. Armor stone must be of sufficient size to withstand the design storm wave condition.



Shore Protection, Figure 1 Photo of a typical bulkhead in Chesapeake Bay, USA. Cross section and plan view of bulkhead construction (Modified from Hardaway and Byrne, 1999).



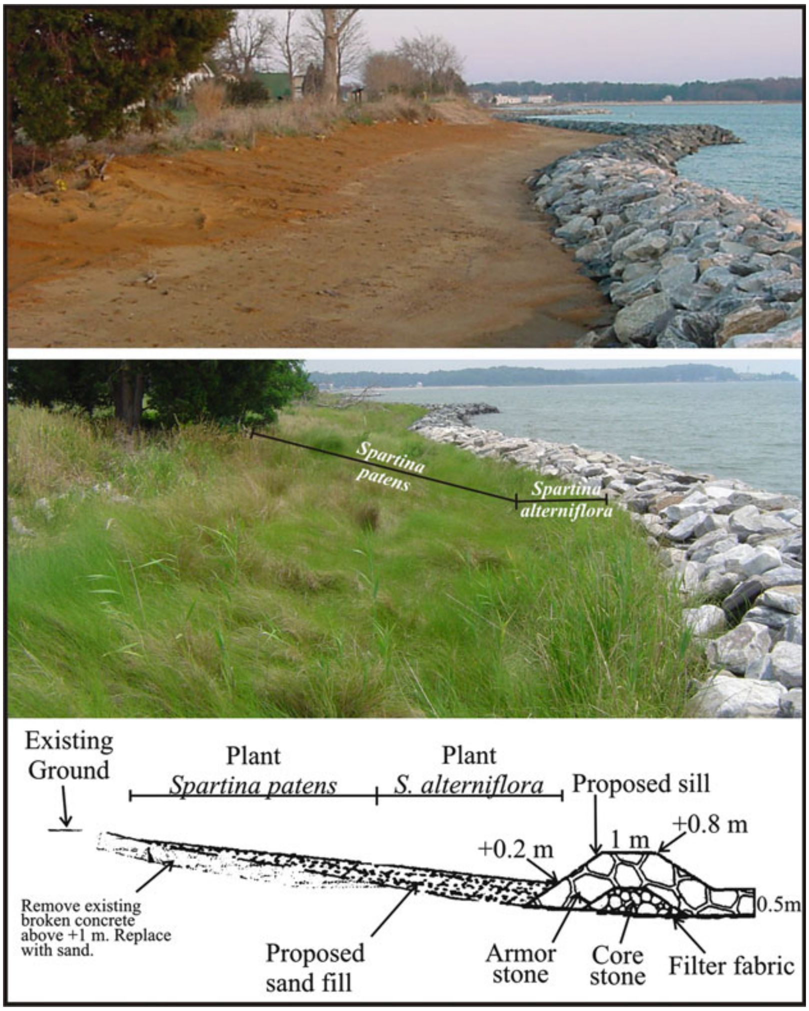
Shore Protection, Figure 2 *Top*: Photo of a typical revetment in Chesapeake Bay, USA, and *bottom*: cross section of elements necessary for proper stone revetment design. Two layers of armor stone overlay a bedding stone layer with filter cloth between the earth subgrade and bedding layer. Armor size is dependent on the design wave height which is determined from an analysis of wave climate for each project site (Modified from Hardaway and Byrne, 1999).

Offensive

Offensive methods include those that extend beyond the eroding bank slope to intercept the impinging wave climate well before impacting the bank. This often involves creating marsh fringes and/or beaches, sometimes referred to as living shorelines. Establishing a marsh fringe or beach for shore protection either by planting the existing substrate or by beach nourishment is considered a nonstructural or “soft” approach. Establishing marsh fringes on their own is limited to very fetch-limited shorelines of <1.6 km (Hardaway et al., 1985).

Marsh fringe establishment in higher fetch conditions (up to 8 km) usually requires the addition of sand and some type of sand containment structure. In the Mid-Atlantic, the use of a sill system is widely used (Figure 3). A sill system consists of sand fill to create stable beach and planting substrate. This is anchored by a rock sill that runs along the nearshore parallel to the coast, where it protects the sand fill from wave scour. Marsh plants are established in the sand fill with intertidal species planted from about mean tide level adjacent to the back of the sill to about mean high water. The sand fill might go up a couple of meters against the eroding bank to a certain level of protection. Bank grading from that point landward increases the stability of the system. The combination of rock, sand, plants and possibly bank grading provides a coastal gradient for storm wave attenuation and long-term sea-level rise (Hardaway et al., 2012).

Establishing a stable protective beach along estuarine shorelines can be done with groins or breakwaters with beach nourishment. Beach nourishment alone is usually only placed on public beaches which seek to create recreational areas in addition to shore protection. When beach nourishment is placed without a structure, ongoing maintenance generally is necessary. Groins often are used to capture littoral sands to create a wide beach.



Shore Protection, Figure 3 Sand fill with stone sill and marsh plantings in Chesapeake Bay, USA. *Top*: after sand fill placement but before planting; *middle*: after 4 years; and *bottom*: the cross section used for construction (From Hardaway et al., 2010).

Worldwide, groins are a commonly used shore protection method. Usually made of wood or rock, the installation of more than one is often referred to as a groin field. Groin length and spacing varies with site conditions. When there is abundant sand moving alongshore, a groin field can be an effective shore protection method (Figure 4). However, in sand-poor systems, downdrift impacts can be significant. In fact, there is almost always an impact downdrift which often leads to a "domino" effect with those properties affected adding more groins.

Offshore breakwaters have been used to provide a stable protective beach in many areas around the world. Along estuarine shorelines, they are either detached or attached breakwaters. Attached breakwaters are often called headland breakwaters. Breakwaters can be used with or without sand nourishment. Though primarily used along open ocean coasts, installations have become more widespread in estuarine setting, particularly Chesapeake Bay (Hardaway and Gunn, 2010, 2011).

Detached breakwater systems (Figure 5) operate on the principal that by placing breakwaters offshore a certain distance that alongshore transport in their lee can continue and thus have minimal impact to the downdrift shoreline. The degree of attachment (or detachment) is primarily a function of breakwater length vs. distance offshore. If the breakwater unit is shorter than its distance offshore, a salient is likely to develop but if the breakwater unit is equal to or closer that its distance offshore, a tombolo will most likely develop (Chasten et al., 1993). Attached breakwater systems (Figure 6) often have a series of breakwater and pocket beaches and usually require sand fill to complete the shore protection system. The sand

Shore Protection, Figure 4 Examples of groin fields in Chesapeake Bay, USA, that have *top*: a sufficient sand supply to create a protective beach and backshore and *bottom*: an insufficient supply of sand such that the groins act as littoral barriers and prevents sand from reaching the downdrift shoreline (From Hardaway and Byrne, 1999).

can be both recreational and protective. Where possible, the sand fill is sloped from the breakwaters to elevation of the desired level of protection at the bank. This provides a sloped area that can reduce the wave energy that impacts the bank under high water conditions (Hardaway et al., 2005).

Design elements

The design of shore protection along estuarine coasts often is predicated on what has been done before and what is locally permissible. In the context of designing an engineered application, consideration of the hydrodynamic forces, usually waves, acting against the shore should be evaluated. Site assessment should include measuring several design parameters: fetch exposure, storm surge frequency, shore orientation, nearshore bathymetry, and bank height and composition. The project wave climate will be affected by these parameters, and the power and frequency storm waves that impact the shore will determine basic components of a shore protection structure such as structure height and rock size. Although developed primarily for open ocean coasts, the Coastal Engineering Manual (USACE, 2000) provides the theoretical basis for physics of shore protection.

The level of protection of a system describes the storm conditions against which a shore protection system would maintain its integrity. In most US localities, the Federal Emergency Management Agency (FEMA) has created

Shore Protection, Figure 5 Detached breakwaters that have a shore salient behind the structures. These structures are located in an area with a strong sand transport system and are designed to allow the sediment to move through the system to downdrift shores.

Shore Protection, Figure 6 Attached headland breakwaters and sand fill create a recreational beach and ecological buffer in an area with an insufficient sand supply. Without structures, any sand placed on the shoreline would be transported away from the site. In order to maintain a sandy beach, the breakwaters are built close to shore so that the sand will maintain an attachment behind the structure minimizing sand loss from the system.

flood insurance rate maps and reports that provide the return frequency of the 10, 50, 100, and 500 years events. These events relate to the annual chance of 10 %, 2 %, 1 %, and 0.2 %, respectively; the mapped area will have a storm surge at that elevation. The elevations associated with the 10 % or 2 % storm events generally are used for design purposes.

According to the NRC (2007), the possibility exists that the level of protection will be exceeded by an event greater than the "design storm." The level of protection employed will translate to the amount of risk or damage the property owner is willing to accept or incur and the amount budgeted for installing protection. Larger projects with more shore protection cost more. Some level of damage may be deemed acceptable depending on the size of the project and the value of the property to be protected.

Conclusion

Ongoing development along the world's estuaries has made shore protection necessary for erosion abatement and the protection of costly infrastructure. The decision to protect estuarine coasts comes down to desired level of protection and costs. Shoreline protection using defensive systems or hardening the coast is a proven commodity. Likewise the widespread use of offensive systems to create beaches and marshes has shown their ability to provide shore protection, if done properly, and should be considered from an estuarine habitat perspective. Over the long term, the broader coastal profile delivered by these systems may allow habitat transition where tides continue to rise.

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Cross-references

Bulkheads
Coastal Erosion Control
Headland Breakwater
Revetments

SHOREBIRDS

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Synonyms

Sandpipers; Waders

Definitions

Shorebirds are small- to medium-sized birds that often feed along shorelines, and they breed in a wide range of habitats near water. They include snipes, godwits, stints, sandpipers, phalaropes, jacanas, thick-knees, oystercatchers, avocets, plovers, and lapwings.

Sandpipers are a group of small familiar shorebirds that often move along the shoreline, foraging by running in and out of the tide.

Waders, another name for shorebirds that is used in Europe, are also called short-legged waders (herons and egrets are long-legged waders).

Introduction

Shorebirds are small- to medium-sized birds that frequently feed along shorelines, often following the waves in and out, picking up prey from the sand. They include snipes, godwits, stints, sandpipers, phalaropes, jacanas, thick-knees, oystercatchers, avocets, plovers, and lapwings. Shorebirds are in 13 families of birds, usually considered in the order Charadriiformes, which also contains gulls, terns, skimmers, and auks (Warnock et al., 2001). Although "shore" is part of their name, they are not generally considered marine birds or seabirds because they do not spend a significant part of their life cycle at sea (Burger, 1984). Phalaropes are the exception; they breed in the prairie pothole region of North American and in the Arctic, migrate to the open ocean, and generally overwinter there before they return to their breeding grounds. Other shorebirds do not wander far offshore, except to pass over the ocean during migration. However, 58 % of shorebirds use marine habitats at some time in their life cycle, and 39 % sometimes or always nest along the coast (Burger, 1984). Because shorebirds travel over large regions of the world during their annual cycle, use a full range of habitats in many biomes and climate zones, and nest in many different habitats, they are ideal sentinels or bioindicators of global environmental change (Piersma and Lindstrom, 2004).

Life history

Shorebirds have intermediate life spans, breed when they are 2–4 years old, usually lay four eggs, have intermediate incubation periods (3–4 weeks), raise one or more chicks a year, and have short parental care periods of less than a month. They have low fecundity, delayed maturity, and high annual survival. In most species, the young are precocial, meaning that they are covered with down upon hatching, and once they dry off, they are able to move about and find their own food, although their parents still guard them until they fledge. The length of the breeding cycle depends somewhat upon where they breed. Species that breed in harsh Arctic environments must have a short period to breed and usually make only one breeding attempt. Species that breed in temperate regions have an extended breeding period and can attempt second broods.

Shorebirds exhibit a range of breeding patterns, from monogamy to polygyny (one male with more than one female mate) and polyandry (one female with more than one male mate; phalaropes). In some species, all three patterns can be exhibited, depending upon food resources. Most shorebirds are territorial when breeding and gregarious when not breeding, often forming large flocks to forage, migrate, and overwinter. Shorebirds have some of the longest migration routes of all birds, with some species breeding in the Arctic and wintering at the southern tip of South America.

Behavior and habitat selection

Breeding

Most shorebirds breed solitarily, although a very few nest in small colonies (e.g., stilts). While nesting, shorebirds rely on their cryptic coloration and hide their nests to avoid mammalian predators. They usually nest on the ground, in the open sand, under vegetation, on rocky beaches, or in wet swales from the Arctic to Antarctica, although some species nest in wet marshes (e.g., snipes), build floating nests in marshes (e.g., jacanas), or nest in sand or salt flats. A very few species, such as the Solitary Sandpiper, *Tringa solitaria*, nest in trees.

Many species are monogamous, maintaining the same mate from year to year, and both members of the pair incubate and care for young. At least 25 species are polygynous, including some sandpipers, snipes, and woodcocks. In a few species, such as Ruff (*Philomachus pugnax*; Van Rhjn, 1991), males gather at a lek (small display and dancing ground) and display to females. The females then go off and breed on their own, incubating without the help of a male. Polyandry occurs in fewer species, perhaps because it is more difficult for the female to lay a complete clutch for different males than it is for a male to fertilize the clutches of several females.

Typically, shorebirds make a nest scrap in the ground and line it with pebbles, shells, grass, or leaves. They lay 1–4 eggs that are pyriform (one end large and rounded, the other small and V shaped) which allows the four eggs to fit nicely together with the four small ends toward the center. Clutch size is a result of their ability to produce the clutch, incubation limitations, and limitations on caring for the young, such as time constraints on high Arctic species because of the short Arctic summer (Colwell, 2010). Both parents incubate, and when the chicks hatch, they leave the nest with the parents within a very few days. The parents lead the chicks to good foraging areas, often along the shore, in muddy swales, or in pools and ponds. Both incubation periods and fledging periods (the time parents continue to guard the chicks, prior to their being able to fly away) relate to body size. In larger shorebirds, the incubation period and fledging period are longer.

Shorebirds are notorious for drawing predators from their nests with a series of distraction displays that involve feigning injury, a broken wing, or an inability to fly, only to fly away when the predator is drawn sufficiently far from the nest. Even some shorebirds that nest in loose colonies (e.g., stilts) will use distraction displays to entice predators to leave the area (Gochfeld, 1984).

Foraging

Shorebirds breed on land and usually winter along the coasts, foraging in estuaries, bays, and along the tide line, but they also make use of saltmarshes, agricultural land, and other upland habitats for roosting or foraging at high tide (Evans-Ogden et al., 2007). They usually locate their prey visually by plucking it from the water, ground, or other surface or by probing in the mud. Morphology affects foraging methods. Shorebird species with larger bills can eat larger prey items, whereas those with longer bills can probe deeper in the sand or mud. Those with longer legs can forage in deeper water (Durell, 2000). On land and along the shore, shorebirds are generally omnivorous, eating a wide range of foods including insects, snails and clams, worms, and other invertebrates, although some eat fish, fruit, seeds, and even carrion. Seedsnipes eat only plant material, while at the other extreme, sheathbills will eat carrion and penguin chicks (Warnock et al., 2001).

Shorebirds are generally solitary during the breeding season, when they forage solitarily as well. Some will even defend foraging territories (Burger and Olla, 1984). During migration, and on the wintering grounds, shorebirds often form foraging and roosting flocks of hundreds to thousands. Particularly dense foraging flocks often occur under circumstances where there is a superabundance of prey that is renewed regularly, as occurs with rising tides. Shorebirds forage and roost in dense flocks as an anti-predator strategy. Foraging in flocks has the advantage of enough eyes to provide early warning of approaching predators, evasive actions of the flock, and predator swamping (whereby any one shorebird has a lower probability of being taken if it is a flock member, rather than being solitary). Predators, such as hawks, however, can have a negative effect on foraging shorebirds in terms of energy cost and lowered survival (Goss-Custard et al., 2006), and they can affect both the timing and the routes of migrating shorebirds (Lank et al., 2003).

Migration and overwintering

Shorebirds breeding in north temperate to Arctic habitats migrate to warmer climates to overwinter, and well over half of the shorebird species migrate. For these species, their strategy is to nest in regions with abundant food supplies during the breeding season, to leave when food is no longer plentiful, and to migrate to regions with abundant food for the winter. They usually follow coastlines or go over the oceans (Morrison, 1984; Warnock et al., 2001). Except for phalaropes, shorebirds rarely touch the ocean surface during migration, and many species spend significant time flying over the oceans. Migration is generally associated with wind patterns, and overwater routes provide energetic savings compared to following the coast. In the advent of light-sensitive global location sensors, it has been possible to document long transoceanic migrations of several species, including nonstop flights of 7,600 km in Ruddy Turnstone (*Arenaria interpres*; Minton et al., 2011). A round-trip migration flight of 26,700 km was reported in Red Knots (*Calidris canutus rufa*), with continuous 6-day flights of 8,000 km (Niles et al., 2010, Figure 1).

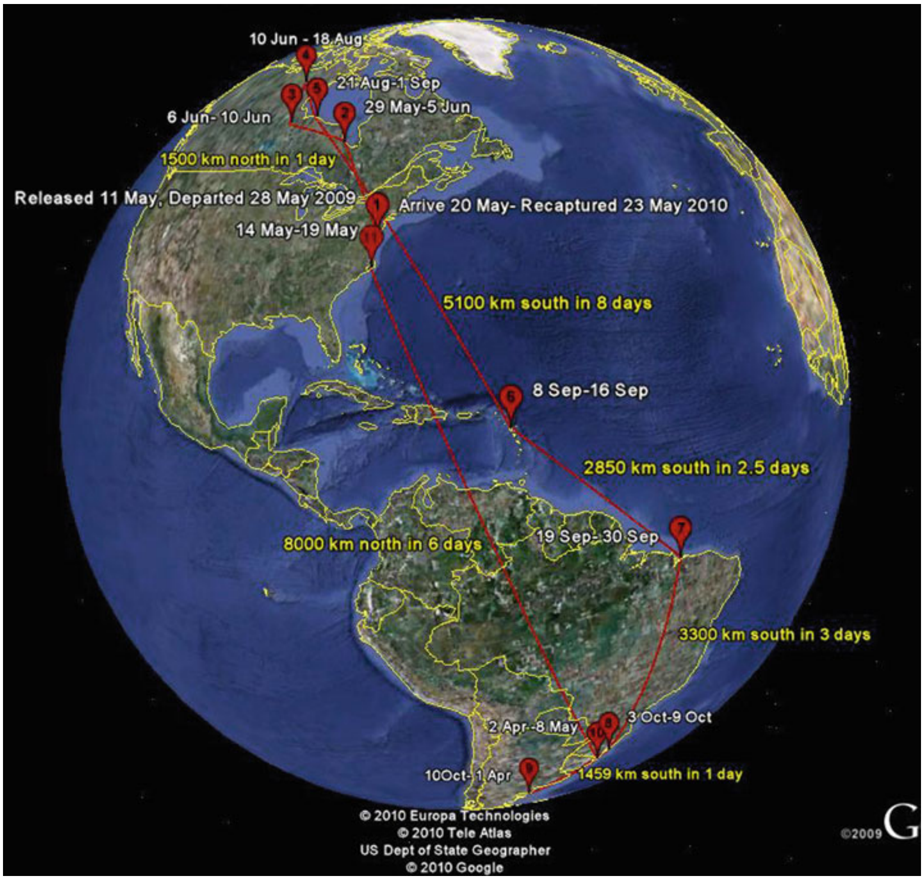
Shorebird populations

Estimates of shorebird populations are difficult to determine because most nest solitarily or in small breeding colonies and are usually scattered throughout the available habitat. Estimates can be determined by counting birds along transects in suitable, known habitats, and by estimating populations on the wintering grounds using aerial counts (Morrison, 2006). Survival rates can be determined with mark-recapture rates, either on the breeding grounds or the wintering grounds (Sandercock, 2003). Determining survival, even for marked birds, is difficult because it is hard separate whether birds survived, simply returned elsewhere, were undetected, or the marking method failed (e.g., bands fell off, batteries died). Understanding population trends of shorebirds, however, requires monitoring over long periods, which has been done for many species in North America (Bart and Johnston, 2012) and Australia (Clemens et al., 2012). An overview of North American shorebirds shows declines in 80 % of 35 species with data (Morrison et al., 2001).

Threats and conservation

At least 21 % of the world's shorebirds (32 of 155) are listed as species of conservation concerns by BirdLife International (Piersma et al., 1997), and this number has increased since then. The main threats to shorebird populations are habitat loss, human disturbance, commercial harvesting of shorebird prey, hunting, pollution, and long-term effects from global warming and sea-level rise (Goss-Custard et al., 2000). While harvesting of the birds may not be a problem in much of the developed world, shorebirds are still harvested in some places, such as South America.

Although Arctic and Antarctic breeding habitat is not generally threatened (except by climate change, oil development, sea-level rise), habitat for breeding, foraging, and migrating shorebirds along temperate and tropical regions is threatened by development, human disturbance, and



Shorebirds, Figure 1 Geolocator output for Red Knot Y0Y: periods when the bird remained in the same location are shown in *white*; the great circle distances of movements are shown in *yellow*. Flight path and stopover location of Red Knot Y0Y. Location key: *1*, Delaware Bay, United States; *2*, James Bay, Canada; *3*, Western Hudson Bay, Canada; *4*, Baker Lake, Canada; *5*, Churchill, Canada; *6*, Lesser Antilles; *7*, Maranhão, Brazil; *8*, Lagoa do Peixe, Brazil; *9*, San Antonio Oeste, Argentina; *10*, Uruguay-Brazil border; *11*, Ocracoke, North Carolina, United States (after Niles et al. 2010).

contaminants (Borgmann, 2011). Habitat for migrating shorebirds is particularly threatened because of the movement of people to coastal regions of the world, and managing habitat for both humans and shorebirds will take collaboration with a range of community stakeholders (Burger and Niles, 2013). Development in Arctic regions, particularly for oil and gas, also provides a threat to shorebird nesting habitat (Kendall et al., 2011).

The vulnerability of migrant and overwintering shorebirds is partly threatened by their migratory patterns and behavior. Many places serve as massive staging and stopover points during migration for thousands of shorebirds, including the Copper River Delta in Alaska, Monomoy Refuge on Cape Cod in Massachusetts, and Delaware Bay in New Jersey, as well as the Wadden (see WHSR, 2013). At such sites, shorebirds are vulnerable not only to habitat loss, human disturbance, and predators but also to food shortages. For example, the massive migration of shorebirds through Delaware Bay in the spring is threatened by depleted horseshoe crab eggs because of declines in the crabs caused by overharvesting by bait fishermen (Niles et al., 2008). Without sufficient crab eggs, shorebirds do not gain enough weight to survive or breed once they reach their Arctic breeding grounds (Morrison et al., 2007). Predators can also affect survival and weight gain during migration (Goss-Custard et al., 2006)

Global warming and sea-level rise provide a long-term threat to shorebirds because of the potential to render breeding, migration, and overwintering habitats unsuitable. Foraging habitat for shorebirds that feed along coasts is estimated to decrease dramatically, even over the next few decades (Convertino et al., 2012). Assuming a conservative global warming scenario of only 2 °C over the next century, Galbraith et al. (2005) predicted that major intertidal habitat losses for shorebirds in bays in Washington, California, Texas, and New Jersey/Delaware would range from 20 % to 70 %. Sea-level rise, however, may be even higher than the initially expected 1 m (Pfeffer et al., 2008). Such massive changes in intertidal areas will affect the amount of suitable foraging habitat for migrating and wintering shorebirds, as well as those breeding along coasts. Changes in the insect populations brought about by global change will also decrease prey abundance and availability (Lindstrom and Agrell, 2012).

Because most species of shorebirds are migratory and often span large geographic regions of the world, their conservation requires international efforts, such as the Western Hemisphere Shorebird Reserve Network (WHSR, 2013). There are two in-depth conservation plans for shorebirds of North America: a US plan (Brown et al., 2001) and a Canadian plan (Donaldson et al., 2000), and both contain a wealth of information on both threats and solutions. These involve collaborations among states, federal governments, international agreements, and treaties.

Summary

Shorebirds, small- to medium-sized birds that frequently feed along shorelines, include snipes, godwits, stints, sandpipers, phalaropes, jacanas, thick-knees, oystercatchers, avocets, plovers, and lapwings. Most shorebirds breed solitarily, although a few nest in small colonies. They rely on their cryptic coloration and hiding their nests to avoid mammalian predation on their nests and eggs. They nest on the ground, in the open sand, under vegetation, on rocky beaches, or in wet swales from the Arctic to Antarctica. During migration and while overwintering, shorebirds form flocks of hundreds to thousands and forage in large groups to exploit coastal prey. Some migrate long distances from Arctic breeding grounds to the tip of South America. Threats to shorebirds include human disturbance while nesting or foraging on coastal beaches and mudflats; predators, harvesting, habitat loss due to coastal and Arctic development, and sea-level rise.

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Cross-references

Climate Change
Coastal Bays
Coastal Wetlands
Deltas
Estuarine Beaches
Food Chain
Food Web/Trophic Dynamics
Habitat Loss
Mean Sea Level
Nonpoint Source Pollution
Oil Pollution
Seabirds
Shoreline
Shoreline Changes

SHORELINE

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Synonyms

Coastline

Definition

Shoreline is defined as the point or line where the sea intersects the land. Considering tidal variations, it corresponds to the mean sea-level position on the shore. The shoreline has shifted in the past with changes in sea level or crustal movements (crustal uplift or sinking).

Description

Due to the glacial eustatic rise in sea level after the last ice age maximum some 20,000 years ago, the shoreline has been changing from about −120 m to the present position. Along many coasts, the past shorelines have shifted due to crustal movements. In seismically active areas, crustal dynamics give rise to sequences of uplifted former shorelines, for example, in Japan (Ota, 1986) and New Zealand (Wellman, 1967). In former glaciated areas such as Fennoscandia, Scotland, and North America, the glacial isostatic process (Jemisson, 1882; De Geer, 1888–1890; Hillaire-Marcel and Fairbridge, 1978; Mörner, 1979, 1980) has tilted the shorelines from the center of glaciation to the periphery. Bravais (1840) was the first to record such tilted shorelines. Gilbert (1890) recorded isostatically deformed shorelines at the former Lake Bonneville in America. There are also submarine shorelines, which appear to represent minor stillstands in the postglacial rise of sea level (see Carter et al., 1986). Sometimes it is possible to isolate the crustal and eustatic components in the spectra of former shorelines such as for the last interglacial–glacial cycle of coral reefs in New Guinea (Chappell et al., 1996) and for the last deglacial phase in Fennoscandia (Mörner, 1971).

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Cross-references

Mean Sea Level
Uplifted Coasts

SHORELINE CHANGES

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Synonyms

Coastline changes

Definition

Shorelines change boundary conditions in an estuary or ocean due to sea level fluctuations, circulation patterns, waves and tides, and the amount of sediment supply. The shoreline is quite narrow, linear in extent, and always in contact with the estuarine waterbody or sea. It is a physical interface of land and an estuary or sea and defined in both temporal and spatial terms.

Introduction

Morphodynamics of a coast can be defined as the “mutual adjustment of topography and fluid dynamics involving sediment transport” (Wright and Thom, 1977) or, alternatively, the “dynamic behaviour of alluvial boundaries” of fluid motions (De Vriend, 1991). The evolution of coastal systems is the result of morphodynamics that develop in response to change in external conditions (Wright and Thom, 1977) and controlled by various factors such as morphology, geology, and size of the catchment area; nature of sediments; climate leading to rainfall and river discharge at coastal zone; freshwater input; and coastal hydrodynamics – waves, tides, and currents (Albert and Jorge, 1998). Many of the regional controls on sea level involve long-term geological processes (subsidence, isostasy) and have a profound influence on controlling short-term dynamics. As sea levels fluctuate, the morphology of a coastal zone will further evolve, changing the boundary conditions of other coastal processes, viz., circulation, waves, tides, and deposition of sediments on shorelines.

The shoreline is a part of coastal land in contact with the estuary or sea and is continuous around ocean basins. It has remarkable linear extent but is usually quite narrow. The width of shorelines is defined by tidal flux – the zone between the lowest low tide and the highest high tide. Because of the dynamic nature of the shoreline boundary, the definition must consider the shoreline in both a temporal and spatial sense (Boak and Turner, 2005). An idealized definition of shoreline is that it coincides with the physical interface of land and water (Dolan et al., 1980). In reality, the shoreline position changes continually through time, because of cross-shore and alongshore sediment movement in the littoral zone and especially because of the dynamic nature of water levels at the coastal boundary (e.g., waves, tides, storm surge, setup, run-up, etc.).

The shoreline is the position of the land–water interface at one instant in time. A shoreline may also be considered over a slightly longer time scale, such as a tidal cycle, where the horizontal/vertical position of the shoreline can vary between centimeters and tens of meters (or more), depending on the beach slope, tidal range, and prevailing wave/weather conditions. Over a longer time scale, such as 100 years or more, the position of the shoreline may vary by hundreds of meters or more (Komar, 1998). The shoreline is a time-dependent phenomenon that may exhibit substantial short-term variability (Morton, 1991).

The shoreline is a vital part of the coastal zone. The study of shorelines is very important to understand the interactions between parts of the hydrosphere, atmosphere, and solid earth. The atmosphere is involved in transferring the energy from wind to water, thereby causing waves, which in turn generate nearshore currents. The gravitational attraction of the moon and sun on ocean waters is responsible for the rhythmic rise and fall of tides. As a dynamic system, shorelines continuously adjust to any change that takes place, such as increased wave energy or an increase or decrease in sediment supply.

Due to the dynamic nature of the shoreline boundary, it can be used as a proxy to represent the “true” shoreline positions. The shoreline positions can be investigated and classified based on (1) visual observation (a previous high-tide line or the wet/dry boundary), (2) tidal datum-based shoreline indicator (mean high water or mean sea level), and (3) application of image processing techniques to extract proxy shoreline features from digital coastal images (Boak and Turner, 2005). Shoreline position measurements of different time periods can be used to derive quantitative estimates of the rate of progradation/retrogradation (Fenster et al., 1993). Human intervention in coastal regions has modified pristine coastlines around the globe by deforestation, cultivation, changes in habitat, urbanization, and upstream obstructions to river flows. The rate of change in coastal landforms and shoreline position is important in the development of setback planning, hazard zoning, erosion/accretion perspectives, sediment budgeting, and conceptual/predictive modeling of coastal morphodynamics (Sherman and Bauer, 1993; Chandramohan et al., 1994; Al Bakri, 1996; Zuzek et al., 2003; Kumar and Jayappa, 2009; Kumar et al., 2010). The discussion below focuses on shoreline processes, erosion and deposition, impact of sea-level rise, and analysis and predictive methods of shoreline changes.

Classification of shorelines

Depositional and erosional coastlines

A sedimentary environment and its sub-environments in a coastal environment can be either erosional or depositional. Erosion, progradation, and reworking of sediments are important shore and nearshore processes that modify coastlines. The details are discussed below.

Submergent and emergent coasts

In some coastal areas where uplift occurs, the sea level falls fast relative to the land. Sea level will rise when the coastal region subsides, resulting in a net change in sea level of as much as 30 cm per century.

Wave-dominated coasts

Wave-dominated coasts, comprising accumulations of detrital sands, undergo high levels of physical reworking interspersed with periods of burial before finally being deposited as present-day coastal deposits. Waves and wave-induced currents are the dominant mechanisms for moving and depositing sands on shorefaces and beaches of the open coast, although winds, river discharge, tidal currents, and Ekman flows act as transporting agents landward of the beach in estuaries and seaward of the shoreface (Roy et al., 1994). In relation to the shoreface and beach, open coastal types are determined by four factors: (1) substrate gradient, (2) wave versus tidal range, (3) sediment supply versus accommodation (Swift and Thorne 1991), and (4) rates of sea-level change. At one end are steep, high-energy, sediment-deficient coasts, and at the other end are low-gradient, low-energy coasts (Roy et al., 1994). The behavior of wave-dominated coasts can be explained under two concepts: (1) geological inheritance or imprint of various land-forming processes that have operated for a long period of geological time and (2) wave-formed coastal deposits operating over shorter periods of time (Roy et al., 1994).

Clastic and carbonate coasts

Clastic coasts are characterized by the relative abundance of river materials and reworked by waves and tides. Carbonate coasts are those where calcareous sediment is produced, transported, deposited, lithified, and eroded. They significantly differ from clastic coasts.

Paleoshorelines

A general history of relative changes in sea level and shoreline migration has been documented on the continental shelves of the world oceans. Relic barrier shorelines of Pleistocene age have been widely reported landward of their modern analogues, while submerged Holocene barrier remnants are common features on modern continental shelves. Barrier shorelines are characteristic features of depositional environments. Barriers can originate from a topographical ridge along the landward side of a beach which may subsequently be partly submerged. The other important characteristic feature of paleoshorelines is cheniers, originally described by Howe et al. (1935) as long, narrow, sandy ridges rising above the surrounding marshes and forming the most conspicuous topographic features along the southwestern Louisiana coast. Cheniers are characterized by gently dipping littoral, sublittoral and washover deposits, and some overlying dunes. Cheniers and barrier shorelines can be identified by the shape and extent of their respective deposits. A fundamental difference in the depositional process is the proximity of a sizeable sediment source which can overload the distributive fores, viz., waves and tidal currents, along the adjacent shoreline.

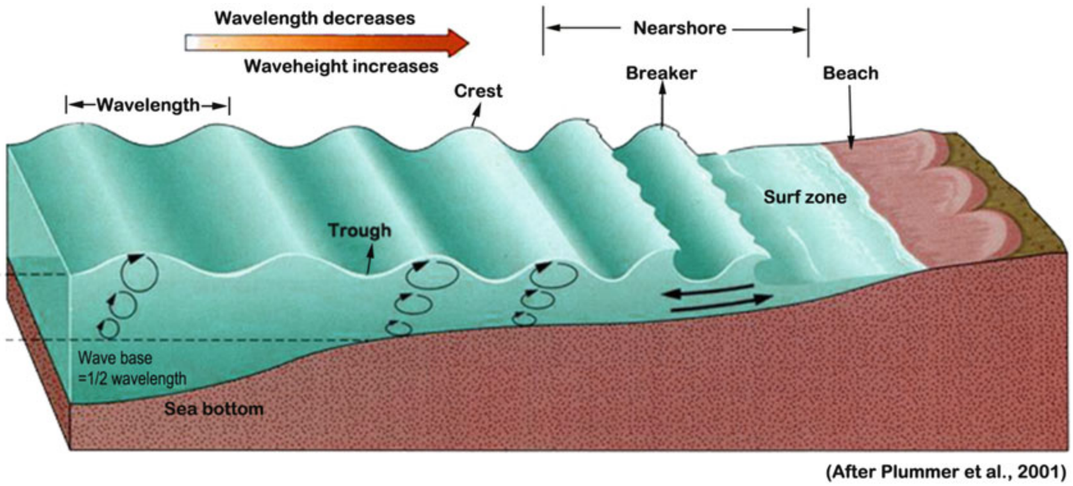
Shoreline changes due to physical forcing

Shoreline processes operate on a narrow zone, and therefore the shoreline migrates landward or seaward depending on changing sea level or uplift or subsidence of coastal regions. These cyclic and noncyclic processes change the position of the shoreline over various time scales, from the daily and seasonal interaction of winds and waves to over thousands of years due to secular sea-level changes. During sea-level transgression, for example, the shoreline migrates landward and vice versa during sea-level regression. Furthermore, shoreline changes are not constant through time and frequently reverse in sign, i.e., accretion to erosion, or vice versa. Most shorelines undergo patterns of erosion and accretion on a daily and seasonal basis and may be unidirectional or cyclic on a long-term basis. In the process, beach and nearshore sediments are deposited over vast regions. However, the physical processes such as tides, waves, and nearshore currents are most important for modifying shorelines.

Tides and tidal currents

In response to the gravitational attraction of the moon and sun, some shorelines experience two almost equal high tides and two low tides each day, called a semidiurnal tide. Some locations experience only one high and one low tide each day (called a diurnal tide), whereas some other locations experience two uneven tides a day or sometimes one high and one low each day (called a mixed tide). These regular fluctuations in the estuarine or ocean surface vary from a few centimeters to >15 m. A complete tidal cycle includes a flood tide that progressively covers more and more of a nearshore area until high tide is reached, followed by ebb tide, during which the nearshore area is once again exposed. These regular fluctuations in sea level constitute one largely untapped source of energy as do waves, currents, and temperature differences in seawater.

Tidal ranges are also affected by shoreline configuration. In offshore areas, where the direction of flow is not restricted by any barriers, the tidal current is rotary; that is, it flows continuously, with the direction changing



Shoreline Changes, Figure 1 Wave dynamics in the nearshore region. Well-developed surf zone and beach are also shown.

through all points of the compass during the tidal period. This rotation, caused by the earth's rotation and unless modified by local conditions, is clockwise in the Northern Hemisphere and counterclockwise in the Southern Hemisphere. In estuaries and/or straits, or where the direction of flow is more or less restricted to certain channels, the tidal current is reversing; that is, it flows alternately in approximately opposite directions with an instant or short period of little or no current (called slack water) at each reversal of the current. During the flow in each direction, the speed varies from zero at the time of slack water to a maximum, called strength of flood or ebb, about midway between the slacks. Tidal currents generally have little modifying effect on shorelines along straight coasts, except in narrow passages where tidal current velocity is strong enough to erode and transport sediment. It works as one of the sediment-transporting agents and to prevent the blockage of passageways created by sediment deposition via the action of nearshore currents.

Several processes such as landslides, earthquakes, and volcanic eruptions in the oceans generate large waves and tsunamis that can devastate coastal areas, but most natural process activities on shorelines are accomplished by wind-generated waves, especially storm waves. Waves are directly or indirectly responsible for most erosion, sediment transport, and deposition in coastal areas.

Shallow-water waves and breakers

Waves are disturbances that cause energy to be transported through a medium (e.g., air or water); they are defined with respect to their height, length, frequency, and wave period (Figure 1). When wind blows over water, the friction generated between the two transfers energy from the wind to the water causing the water surface to oscillate. When waves move across a water surface, the water moves in circular orbits but shows little or no net forward movement. Only the waveform moves forward, and as such it transfers energy in the direction of wave movement. The diameters of the orbits that water follows in waves diminish rapidly with depth, and at a depth of about one-half wavelength (L/2), called the wave base, they are essentially zero. Thus, at a depth exceeding wave base, the water and seafloor are unaffected by surface waves (Figure 1).

When a deepwater wave approaches the shore and moves into shallow water, the resulting friction and compression reduce the forward speed of the wave. Therefore, when the wave "feels the sea bottom," it slows down, and the accompanying reduction in the wavelength and speed results in increased height and steepness as the wave energy is condensed in a smaller water volume. The influence of depth on the propagation of waves increases with continued shoaling until it becomes the dominating factor.

Breakers form in the surf zone because the water particle motion at depth is affected by the bottom. Orbital motion is slowed and compressed vertically, but the orbit speed of water particles near the crest of the wave will not slow down appreciably. The particles at the wave crest move faster toward the shore than the rest of the wave form, resulting in the curling of the crest and the eventual breaking of the wave. The two most common types of breakers are plungers and spillers. Spilling breakers move forward with a foaming turbulent crest, while plunging breakers form on narrow, steep beach slopes. The more common spilling breakers are found over wider, flatter beaches, where the energy is extracted more gradually as the wave moves over the shallow bottom. The spilling breakers last longer than the plungers, because they lose energy more gradually.

Nearshore currents

Wave action in and near the breaker zone carries mass transport of water shoreward as longshore currents, rip currents, and the longshore movement of the expanding heads of rip currents. In the nearshore zone, where the breaker zone and surf zone are located, the water from breaking waves rushes forward and then flows seaward as backwash. Longshore currents are one important aspect of breaking waves, and these waves have the ability to generate currents that flow parallel to the shoreline. They occur because each wave thrusts the water forward when it breaks. Longshore currents are long and narrow, and they flow in the same general direction as the approaching waves. Longshore currents are particularly important in transporting and depositing sediment in the nearshore zone. Rivers are one of the major sources of littoral drift and the annual discharge of sediments to the seas. Along the Indian coasts alone, the annual sediment discharge to the ocean is about 1.2 billion tonnes, which accounts roughly 10 % of the global sediment flux to the world oceans (Subramanian, 1993).

Monsoons play an important role in shoreline changes and configurations, particularly along Asia and Southeast Asian seas. The Indian coast experiences two monsoons – the southwest (summer) monsoon (June–September) and the weaker northeast (winter) monsoon (October–December). During the southwest monsoon, the coastal current is in a clockwise direction, while during the northeast monsoon, the current is in a counterclockwise direction. As a result, the longshore current is stronger and toward the south during the southwest monsoon (Narayana et al., 2001). Natural processes such as waves, littoral currents, offshore relief, rainfall, and sea-level changes are responsible for erosion/accretion of the coast and subsequent shoreline changes. Surging, spilling, and plunging breakers with wave heights of 2–3 m occur along the monsoon-dominated coasts. In the monsoon season, wind energy is much greater, resulting in larger amplitude waves and strong littoral currents. Infragravity and far infragravity edge waves, coupled with strong reflections and undertow, play an important role in the hydrodynamics of the coastline (Tatavarti et al., 1996). Sediments lying on the nearshore bed are disturbed and eroded during the monsoon season because of the larger and strong waves and undertow. Therefore, sediments are mobilized into suspension and transported. As the low-frequency motions are three-dimensional, they carry suspensions laterally (Tatavarti et al., 1999), thereby resulting in variations in shoreline positions. Further, whenever the wind activity is strong, well-formed cusps with spacing of 15–20 m alongshore with vertical amplitudes of about of 0.5 m are formed (Figure 2).

Shoreline Changes, Figure 2 Gentle to moderate beach slope with beach cusps in the background, near Kotepura, west coast of India (Photo: K.S. Jayappa).

Deposition along shorelines

Depositional coasts are steady or growing because of their sediment accumulation rate. Rivers are one of the major sources of sediment supply and littoral drift of sediments in the coastal seas. Net sediment transport direction can be inferred on the basis of accumulation of sediment on the updrift and erosion on the downdrift directions of breakwaters, shifting of river mouths, length of drift sectors, and littoral sediment transport direction. Wave refraction and the resulting longshore currents are the primary agents of sediment transport and deposition on shorelines. Tides also play a role as they rise and fall, and the position of wave attack shifts onshore and offshore. Rip currents play no role in shoreline deposition, but they do transport fine-grained sediments (silt and clay) offshore through the breaker zone. The rate of littoral sediment transport depends on the angle of wave approach, wave energy, intensity of longshore currents, and sediment supply. There is a well-established view that the net littoral transport rate along the beaches ranges from near zero to 7.65×10^5 m^3/year (Shore Protection Manual, 1984). Littoral transport determines the areas of coastal erosion and deposition, and it influences the morphology, orientation of coastal landforms, and evolution of the coast.

Mudbanks (fluid muds) generally occur on coastlines supplied with large quantities of river-discharged muds which advect and diffuse across and along the shelf and nearshore environment. Coastlines such as Surinam and French Guiana are dominated by river-derived suspended mud along foreshore, shoreface, and shelf. However, the mudbanks of the southwest coast of India are not associated with river mouths but occur as discrete features on otherwise sandy shorelines. These mudbanks are associated with attenuated nearshore wave energy and high suspended-sediment concentrations during and for some time after the summer monsoon period (June–September). Their impact on wave energy minimizes coastal erosion and provides extensive deposition along the shore in the mudbank/fluid mud regimes.

Extreme events such as tsunamis also contribute to extensive deposition of sediments on shorelines (Narayana et al., 2007).

Shoreline Changes, Figure 3 Cliffed and irregular shoreline protected with seawall near Kannur, southwest coast of India (Photo: Avinash Kumar).

Shoreline erosion

Coastal erosion is generally related to wave energy, shoreline material, coastal topography, and the direction of the approaching waves with respect to the shoreline direction. The breaking waves and currents in the nearshore zone are responsible for the transport of coastal sediments resulting in shoreline change. The first effect of erosion on a newly exposed coast is intensification of the coastline's irregularity (Figure 3). In the long run, shoreline processes tend to straighten an initially irregular shoreline. Wave refraction causes more wave energy to be expended on headlands and less on embayments. Thus, headlands erode, and some of the sediment yielded by erosion is deposited in the embayments.

The shoreline changes in India suggest that erosion/accretion is cyclic. Beach width reduces between 15 and 50 m during June and August because of intense monsoonal erosion, whereas beaches attain maximum width during February–April because of accretion along the coasts like Western India. The intensive tropical monsoons cause large-scale shoreline erosion. However, the sand that was lost during the monsoon is regained and accreted during the post-monsoon periods.

Erosional coasts are those in which the dominant processes remove coastal material. The nature of beach material plays an important role in modifying the characteristics of incoming waves. This is a matter of prime concern with regard to the character of the beach because the resistance of beach sands to erosive forces depends on particle size. Erosion creates steep or vertical slopes known as sea cliffs. Globally, about 80 % of the open coast is backed by sea cliffs (Bird, 2000). Wherever the rocky shores containing weakly resistant sedimentary units – "soft-rock cliffs" – are exposed to wave action, they have a tendency to be unstable and to rapidly retreat. Coastal cliffs generally form by undercutting due to marine erosion followed by subsequent collapse of large rocky boulders (Woodroffe, 2002). Predictions of coastal cliff recession are essential for an appraisal of cliff protection options and for coastal land-use planning (Hall et al., 2000).

Erosive forces can produce wave-cut shores (Figure 4). Wave intensity and the resistance of shoreline materials to erosion determine the rate at which a sea cliff or shoreline retreats landward. Due to hydraulic action and abrasion at their bases, sea cliffs slope abruptly from land into the ocean, their steepness usually resulting from the collapse of undercut notches. Thus, sea cliffs retreat gradually and leave a beveled surface called a wave-cut platform that slopes gently seaward. Wave-cut platforms above sea level are known as marine terraces.

Sea cliffs generally retreat irregularly because some of the constituent material is more resistant to erosion than other material. Headlands are seaward-projecting areas of the shoreline, eroded on both sides by wave refraction (Figure 5).

Shoreline changes due to natural processes

Sandy shorelines are generally dynamic and exhibit temporal and spatial changes. Sediment supply, littoral drift, and secular sea-level changes are the main factors that influence shoreline changes and the formation of different coastal landforms, while river flow and wave breakers play a significant role in shaping and orientating them (Kunte and Wagle, 1991; Narayana and Priju, 2006). The shoreline configuration is influenced by an accelerated or decelerated accretion of sediment. Accelerated

Shoreline Changes, Figure 4 Cliffed shoreline cut by monsoonal waves.

Shoreline Changes, Figure 5 Irregularly retreating shoreline. Headland of the shoreline projecting seaward is eroded on both sides by wave refraction.

accretion or decelerated erosion results from greater sediment deposition, whereas decelerated accretion or accelerated erosion suggests greater sediment transport (Morton, 1979). Deposition and erosion of beach sediments depend on shoreline configuration, source and sink of sediment, and the hydrodynamics of the nearshore region. Although the overall direction of sediment transport and its mechanisms determine the areas of coastal erosion and accumulation, factors such as intensity of monsoons influence the erosion and accretion patterns along the west coast of India (Narayana et al., 2001). Therefore, it is important to understand how sediments from various sources on the beaches are reworked and redistributed by the nearshore hydrodynamic processes.

Shoreline Changes, Figure 6 Wave-cut notches and erosion on the downdrift of a river mouth in Karnataka, India (Photo: K.S. Jayappa).

A strong relationship was reported between the variability of rainfall and sediment transport, where high sediment discharges are recorded with high rainfall (Syvitski and Morehead, 1999). Further, intensive monsoons make the sea rough, with high wave activity, and erosion of the sediment along the coast, resulting in change of the shoreline configuration. In summary, intensive monsoon rainfall and sediment derived from inland areas influence the configuration and position of the shoreline. The oceanographic regime is dominated by meteorological forcing rather than tidal forcing along the Asian coasts, where tidal range is ~1 m. If the accumulation of sediment at the updrift arm completely balances the erosion at the downdrift end, a "straight" inlet develops. A curved spit, projecting toward the inlet, suggests that spits display marked changes in form and alignment in response to relative sea-level variations, sediment supply, and wave climate (Firth et al., 1995).

Recession of cliffed coasts is the cumulative result of a number of interacting forces and activities. It can be measured or estimated from identified, common, or analogous cliff features and sequentially plotted over the longest possible time periods (Malcolm and Janet, 1997). When waves attack a permeable cliff base, notches of various shapes develop depending upon the wave conditions (Figure 6). Formation of wave-cut notches on the lower part of cliff faces leads to collapse of the upper part of the entire cliff and rapid retreat. In the case of a high cliff, the rate of retreat is even greater because the overlying weight exerts pressure on the roof of the notches. For a cliff of intermediate height, wave-cut notches neither reach the cliff edge nor induce collapse as rapidly as in a high cliff, and the metastable profile can be maintained longer. Various mechanisms leading to detachment of materials from the parent rock include mass movement, seepage erosion, surface erosion (rain flash and wind erosion), and wave attack (abrasion, hydraulic action, and fluid shear by uprushing waves during large storms) (Sunamura, 1991). The presence of cap rocks facilitates multiple rotational slides (Bromhead, 1979) that characteristically produce high-magnitude but low-frequency recession events (Brunsden and Jones, 1980). Groundwater reservoirs confined to permeable strata that overlie or interbed with impermeable units produce seepage erosion at the cliff face (Hutchinson et al., 1981) and also facilitate major mass movements (Denness, 1971).

Impact of sea-level rise (SLR)

Sea-level rise during the late Pleistocene and Holocene dramatically altered the physiography of the coastlines around the world. With the onset of sea-level rise, around ~18 ka, the coastlines began to migrate landward (Vanderburgh et al., 2010). The high rate of sea-level rise during early to mid-Holocene time, high sediment discharge, and wave energy regime favored the preservation of transgressive depositional sequences. A reduction in the rate of sea-level rise ~3 ka, with a subsequent stabilization of sea level ~2.4 ka, resulted in a change from transgression to regression in most coastal regions of the world, as reported by Vanderburgh et al. (2010) for Columbia River littoral zone.

Sea-level rise (SLR) today has been largely attributed to global warming. Global warming has added water to the oceans by melting ice in the polar regions, but the greater contributor is thought to be thermal expansion of the oceans, a rise in sea level due to increasing water temperature. It has vulnerable and direct impact on coastal communities (~10 % of population lives within an elevation of 10 m above mean sea level) (McGranahan et al., 2007). Although sea level has been rising since the end of the last glaciation (nearly 11,000 years), the rate of sea-level rise has increased over the past 200 years as average temperatures have increased. Sea level has risen 10–25 cm in the past 100 years, and it is predicted to rise another 50 cm over the next century. Tide gauge data indicate that the global sea level has risen, on average, by 1.5–2.0 mm/year in the last century (Miller and Douglas, 2004), and since 1993, the rate has increased to 3 mm/year (Church and White, 2006). There are large regional variations of sea level (Cazenave and Llovel, 2010). Since 2003, the mean rate of global sea-level rise has declined to 2.5 ± 0.4 mm/year (Ablain et al., 2009).

The absolute rate of sea-level rise in a region is mainly due to two factors: (1) the increase in volume of water in the ocean basins as a result of increasing glacial ice melting and (2) the thermal expansion of near-surface seawater (Milne et al., 2009; Stammer et al., 2013). Several studies indicate that eustatic sea level will continue to rise because of global warming caused by increasing concentrations of greenhouse gases in the atmosphere. More locally other processes must be considered including vertical land motions such as subsidence or uplift due to tectonic and volcanic activity, subsidence due to sediment loading, ground water pumping, and oil and gas extraction (Woppelmann et al., 2007).

The IPCC Report suggests that the global sea level will be ~600 mm by 2100 AD. This means the annual rate of increase would be 6.45 mm. Furthermore, if the ice caps continue to melt, there could be a 1 m rise of eustatic sea level by the end of the twenty-first century (Pfeffer et al., 2008). Sea-level rise has a direct impact on shoreline changes due to a higher shift in the zone of wave action on the beach. This would lead to shoreline recession, which will be larger on gentler slopes. Bruun (1962) developed a model which estimates shoreline recession with respect to rise in sea level. The effect of sea-level rise will be manifested by greater erosion of beaches and bluffs, increased flooding, inundation of low-lying areas, intrusion of salt water into aquifers, and higher water tables (Gornitz, 1991; Nicholls and Leatherman, 1995). Sea-level rise will strongly impact most coastal landforms (e.g., beaches, lagoons, estuaries, deltas, coral reefs, mangroves, etc.), but the impacts would be spatially variable depending on local factors. Low-lying areas of developing countries are likely to be the most greatly impacted (Nicholls et al., 2007).

Shoreline changes due to anthropogenic activities

Anthropogenic activities such as construction of coastal structures (harbors, breakwaters, seawalls, and vented dams across rivers), mining of sand and shells, and urbanization and industrialization mainly affect the shoreline configuration and coastal morphology. Bulkhead- and revetment-type seawalls have been built along the eroding shores. Sandbags and gabions have also been used in these erosion-prone areas (Figure 7). However, they can induce severe erosion in several locations along the coast and increase the beach slope in front of the seawall as well as on adjacent beaches. Seawalls often produce rubble along the coast and degrade recreational beaches with episodic damages. Groin trap littoral drift resulting in the accumulation of sediment on the updrift side and erosion on the downdrift side. A large quantity of sediment is arrested by these structures, causing a deficit in sediment supply along the coast.

Extensive mining of sand and lime shells in the estuaries and river mouths also leads to accelerated erosion along the coastline. Reduction or loss in supply of sediment affects the dynamic equilibrium of beaches and hastens erosion. In the last few decades, rapid urbanization and industrialization such as construction of houses, fish-processing units, major oil refineries, etc., have accelerated changes in the coastal region. Tourism and recreation are other human activities that often damage natural vegetation, which can enhance coastal erosion and change shoreline configuration.

Shoreline-change analysis and predictions methods

Research on coastal changes provides important environmental indicators for coastal management (Welch et al., 1992; Stokkom et al., 1993). Coastal mapping methods are valuable tools to understand shoreline changes. Shoreline mapping provides critical shoreline data for models used to represent shorelines in the geographic database. Shoreline-change analysis methods can be applied. By knowing the data acquisition method, the inherent errors that normally exist in the underlying measurement processes can be identified and modeled. Also, by knowing the models used to represent shorelines in the geographic database, the level of abstraction of the real world inherent in these models can be recognized. This directly influences shoreline-change analysis results.

The most effective and economic instruments used in shoreline mapping and shoreline-change monitoring are satellite sensors, Global Positioning Systems (GPS), and all-weather sensors (Li et al., 2001). Shorelines can be extracted from the stereo-matched and geo-referenced aerial photographs both manually and automatically. The manual extraction of shoreline features is a process that involves digitizing the water and land interface, which is known as the instantaneous shoreline at the time of aerial photography. The automatic shoreline extraction process

Shoreline Changes, Figure 7 Fury of the monsoon waves slashing the coastline with intensive erosion. Surf zone between the wave break and shoreline is well developed. Seawall is built as a protective feature (Photo: K.S. Jayappa).

involves the classification of the gray values in the processed aerial photographs to obtain the water and land interface. Shoreline-change studies using remote sensing techniques are highly accurate and cost-effective. Satellite-imaging systems have increasingly improved image resolution; the new generation of high-resolution satellite imagery, such as QuickBird and IKONOS which has a resolution of 1 m with stereo imaging capability, provides an example (Fritz, 1996; Li, 1998). An investigation of shoreline mapping using such high-resolution satellite images demonstrates a promising mapping accuracy of 2 m and a great reduction in the number of ground control points required (Li et al., 2001).

Shoreline-change analysis

Shoreline change can be accurately evaluated by subdividing the shoreline into smaller segments by creating transects at right angles to a master shoreline. Shoreline changes along the transects can be computed and further used to predict future shoreline changes (Carter, 1986; Kumar et al., 2010). This method has been adopted over the years to establish the correspondence between shoreline models acquired at different times to predict shoreline change (Fenster et al., 1993; Maiti and Bhattacharya, 2009; Kumar et al., 2010). Rates of change are then employed to summarize historical shoreline movements and to predict future positions based on the perceived historical trends. The method commonly used especially by coastal land planners and managers to predict future shoreline changes is an extrapolation of a constant rate of change (Owens, 1985; Deepika et al., 2013). This method makes use of successive shoreline data available over time, which enables assessment of future shoreline changes by reviewing the spatiotemporal changes of the shoreline.

Shoreline-change rate calculation methods

Due to the shifting of shoreline position and human influences on coastal processes and sediment sources, it is critical to determine whether the long- or short-term rates of shoreline change reflect present-day shoreline dynamics. This analysis may be complicated in areas that exhibit trend reversals (erosion to accretion, and vice versa), or where human activities, such as revetment construction, have affected sediment sources and altered shoreline processes. An understanding and proper application of short-term shoreline changes and long-term data are critical components for effective shoreline management. Professional judgment and knowledge of natural and human impacts are essential in determining whether the long- or short-term data should be used for management purposes particularly in areas that exhibit significant or frequent shoreline trend reversals or areas that have been extensively altered by human activities.

The shoreline-change calculation and prediction techniques allow the stability of a long-term trend relative to intermediate (>50 years) and short-term (decennial) trend, thereby relating the past with the expected future shoreline positions. This section describes the various statistical methods used to calculate shoreline-change data, as well as the methodology used to generate the baseline and transect locations. Various methods of determining shoreline rates of change have been described by Dolan et al. (1991) and Kumar et al. (2010), which are widely considered the definitive works on the subject. All methods used for calculating shoreline rates of change involve measuring the differences between shoreline positions through time. Rates of shoreline change are expressed in terms of distance of change per year.

Methods, such as end-point rate, average of rates, linear regression, jackknife, and average of eras rates, are adapted in estimation of shoreline changes.

End-point rate (EPR)

End-point rate can be calculated by dividing from the two end points – the earliest and latest positions – the distance of shoreline movement by the time elapsed between the earliest and latest measurements at each transect. The future shoreline position for a given date can be estimated using the rate and intercept (Fenster et al., 1993):

$$\mathrm{Y} = \mathrm{mX} + \mathrm{B} \tag{1}$$

where Y denotes the shoreline position, X the date, B the intercept, and m the rate of shoreline movement. Given shoreline data sets, numbered in ascending order by date, the EPR intercept is

$$\mathrm{B}_{\mathrm{EPR}} = \mathrm{Y}_{\mathrm{n}} - \mathrm{m}_{\mathrm{EPR}} \times \mathrm{X}_{\mathrm{n}} \tag{2}$$

Average of rates (AOR)

This method involves calculating separate end-point rates for all combinations of shorelines when more than two are available and can be extended to incorporate the accuracy of the shoreline position data and the magnitude of the rate of change. Foster and Savage (1989) developed an equation to determine if any given EPR meets a minimum time criterion ($\mathrm{T}_{\mathrm{min}}$):

$$\mathrm{T}_{\mathrm{min}} = \frac{\sqrt{(\mathrm{E}_1)^2 + (\mathrm{E}_2)^2}}{\mathrm{R}_1} \tag{3}$$

where E_1 and E_2 are the measurement errors in the first and second point, respectively, and R_1 is the EPR of the longest time span for a particular transect.

The advantages of using AOR are that all the EPRs that survive the minimum time span equation are used and it allows calculation of the time-dependent variance from the average of rates. The two primary disadvantages of using AOR to compute long-term trend are as follows: (1) there is no computational norm for modeling the minimum time span equation, and (2) the results are sensitive to the choice of values selected to represent the measurement errors (Dolan et al., 1991). Foster and Savage (1989) do not recommend AOR as a general computational method, but it can be used as a method of verification in combination with EPR and LR.

Linear regression (LR)

A linear regression rate-of-change statistic can be determined by fitting a least squares regression line to all shoreline points for a particular transect. LR is the most reliable method to predict future shoreline positions and their associated confidence intervals, if measurement errors and a linear trend of erosion were the only determining factors over the longest possible period of shoreline position (Crowell et al., 1997; Douglas and Crowell, 2000). Linear regression can reveal if an association exists, and in particular (via the *r* value) what fraction of the variance of the dependent variable (shoreline position) is attributable to the independent variable (time). This method uses all the available data from many data sets to find a line, which has the overall minimum of the squared distance to the known shoreline.

To calculate the rate of change and to predict the future shoreline position using linear regression, the oldest shoreline position is chosen as a baseline or zero (0) position to measure the amount of shoreline shift (Maiti and Bhattacharya, 2009; Kumar et al., 2010). With reference to this baseline, progradation of the shoreline is considered as a positive value, and recession is considered as a negative value. The change in shoreline position rate is calculated by the linear regression equation $y = \alpha + \beta t$, where y is the shoreline shift during the year t, with $y = 0$ for $t = X$. The regression coefficient (β) represents shoreline-change rate, and the correlation coefficient (*r*) is a measure of goodness of fit of the equation to the present data. Based on the number of samples, the statistical significance is to be considered at the 80 % level of confidence (if number of samples is small) and 95 % confidence level in the case of large numbers of samples as suggested by Allan et al., (2003).

Jackknife

The jackknife method is used as an iterative linear regression that calculates a linear regression fit to shoreline data points with all possible combinations of shoreline points, leaving out one point in each iteration. The slopes of the linear regression lines are averaged to yield the jackknife rate. The advantages of the jackknife are similar to linear regression; the jackknife is also less influenced by outliers of data clusters. The main disadvantage of the jackknife is a lack of increased statistical value given the typically small numbers of shoreline data points used to derive a shoreline rate of change. The statistical utility of the jackknife is best realized with an order of magnitude of (more) data points.

Summary

Shoreline, a part of coastal land in contact with estuarine or ocean waters, undergoes various morphodynamical changes. The shorelines migrate landward or seaward depending on changing sea level or uplift or subsidence of coastal regions. The cyclic and noncyclic processes change the position of shorelines over time scales, from the daily and seasonal interaction of winds and waves to over thousands of years due to secular sea-level changes.

Tidal ranges affect the shoreline configuration, and vice versa. Monsoons also play an important role in shoreline changes. Wave refractions and the resulting longshore currents are the primary agents of sediment transport and deposition on shorelines. Rivers are the major sources of sediment supply and littoral drift of sediments along shorelines. The breaking waves and currents in the nearshore zone are responsible for the transport of coastal sediments resulting in shoreline change. Anthropogenic activities such as construction of coastal structures (harbors, breakwaters, seawalls, and vented dams across the rivers), mining of sand and shells, and urbanization and industrialization also contribute to the shoreline changes.

Sea-level rise during the late Pleistocene and Holocene dramatically altered the physiography of the coastlines around the world. The high rate of sea-level rise during early to mid-Holocene time, high sediment discharge, and wave energy regime favored the preservation of transgressive depositional sequences.

Shoreline-change analysis is very important to understand coastal processes and morphodynamics, as well as to predict the future shoreline changes. The most effective and economic instrumentations used for shoreline mapping and shoreline-change monitoring are satellite sensors, Global Positioning Systems (GPS), and all-weather sensors. Shoreline change studies using remote sensing techniques are highly accurate and cost-effective. Satellite-imaging systems have increasingly improved image resolution including the new generation of the high-resolution satellite imagery such as QuickBird and IKONOS. Various methods, such as end-point rate, average of rates, linear regression, jackknife, and average of eras rates, are used to estimate shoreline changes.

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Cross-references

Anthropogenic Impacts
Beach Management
Beach Processes
Estuarine Beaches
Shoreline

SHORELINE UNDULATIONS

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Synonyms

Shoreline sand waves

Definition

Shoreline undulations are medium to large spatial scale shoreline geomorphic features with longshore dimensions ranging from hundreds to thousands of meters and amplitudes from tens to hundreds of meters.

Description

Shoreline undulations are generally classified under rhythmic coastline features, although many examples are neither periodic nor regularly spaced (López-Ruiz et al., 2012). They can be episodically or persistently found along many shorelines all over the world, including regular rectilinear or slightly curved beaches and river mouths and estuaries. They are also frequently found associated with sudden changes in the orientation of the coast such as at spits (Kaergaard and Fredsoe, 2013) and near human infrastructures. Many authors refer to these features as shoreline sand waves, but the latter are generally considered rhythmically spaced and migrating alongshore. It is generally assumed that, when an undulation is present at the shoreline, the bathymetry is parallel to this undulation down to a certain depth.

The mechanism(s) behind the formation of shoreline undulations is still not well understood. The main working hypothesis in recent years is that coastlines with a wave climate dominated by very oblique incidence commonly feature large-scale undulations, suggesting that this instability mechanism could be mainly responsible for the formation of regularly spaced shoreline undulations. Recent advances reveal that the variation of alongshore sediment transport with the angle formed by the wave crests and the coastline, as well as the surf zone width, plays a major role in the development of shoreline undulations. This process is important at estuarine and river mouths, littoral spits, and curved coastlines and close to human interventions (e.g., jetties) where the nearshore wave regimen is intensively modified. Whatever the case may be, detailed description of the nearshore hydrodynamics is required for modeling shoreline undulations.

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Cross-references

Coastal Barriers
Coastal Landforms
Sand Ridge
Sediment Transport
Shoreline
Spit

SIDE-SCAN SONAR IMAGING OF SEDIMENT BEDLOAD

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Synonyms

Side-imaging sonar of bottom sediments; Side-scan sonar of bed morphology; Side-scan sonar of sedimentary beds

Definition

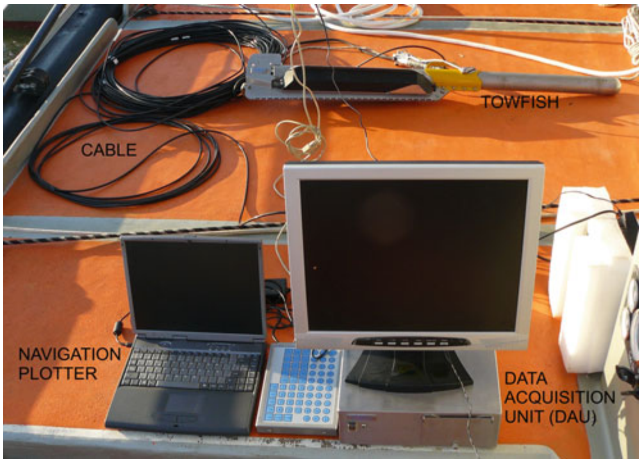
Side-scan sonar is a type of acoustic sonar system used to obtain clear images of the surface of underwater floors. The system works by means of a beam of acoustic pulses which open vertically as a fan shape to both sides of the ship. Each acoustic pulse reflects on the floor and returns to the sensor. The intensity of the received reflected pulse depends on the nature and morphology of the bottom. The conjunction of all the received pulses can build an accurate image of the bed.

Sediment bedload refers to coarse sedimentary particles like sand and gravel that are transported very close to the bed because the relationship between the grain size and flow velocity is near the transport threshold.

Introduction

Side-scan sonar imaging is a geophysical method used in marine engineering, underwater archaeology, and military applications (mining detection). It is also used for sedimentological interpretations of bed configurations (Wright et al., 1987; Anthony and Leth, 2002; Kuijpers et al., 2002; Gómez et al., 2010). In this case, the application of the acoustic geophysical techniques enables the study of the sedimentary dynamics in estuaries and coastal marine areas. The use of side-scan sonar is vital in estuarine research because it allows the rapid scanning of large areas of the estuarine floor to yield images or records for interpretation of the beds.

Side-scan sonar uses acoustic pulses of frequency that ranges from 100 to 1,000 kHz (Blondel, 2009) emitted from transductors located in a submerged towfish that is connected by a cable to a surface vessel (Figure 1). The acoustic pulses are reflected in the bed and return to the receptors also located in the towfish (Figure 2). The towfish sends the received information to the vessel unit, which processes the information to be transformed in an image of the bed. The intensity of the acoustic response of the floor depends on the nature (reflectivity and texture) of the bed material and the orientation that presents the bed surface with respect to the acoustic pulse. The correct interpretation of the records requires an accurate positioning of the images. For that, the system is normally connected with a GPS, obtaining a geo-referenced position of each point of the recorded image. A systematic navigation allows the scanning of wide underwater surfaces, and the accurate geo-position of successive records is the base to build geo-referenced photomosaics of the estuarine bed. Images obtained by side-scan sonar can be as precise as a photograph of the bed and clearly reveal sedimentary features and bedforms.

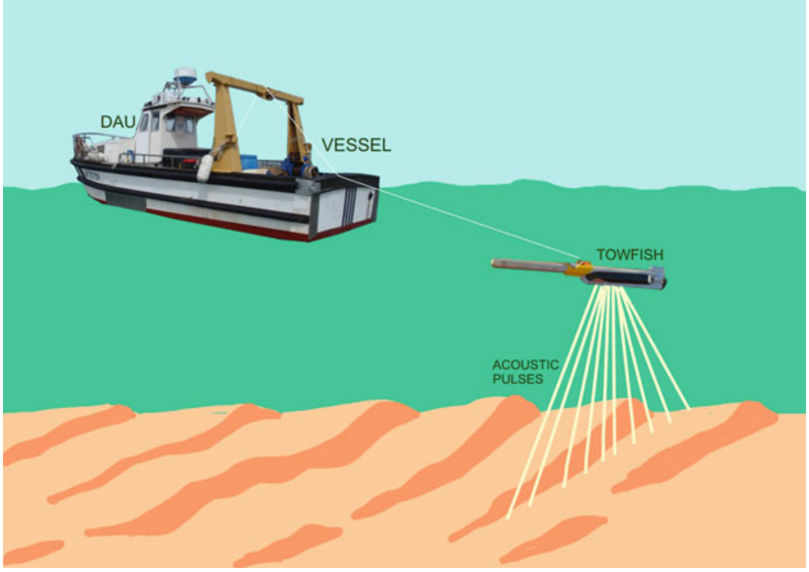


Side-Scan Sonar Imaging of Sediment Bedload, Figure 1 Side-scan sonar equipment.

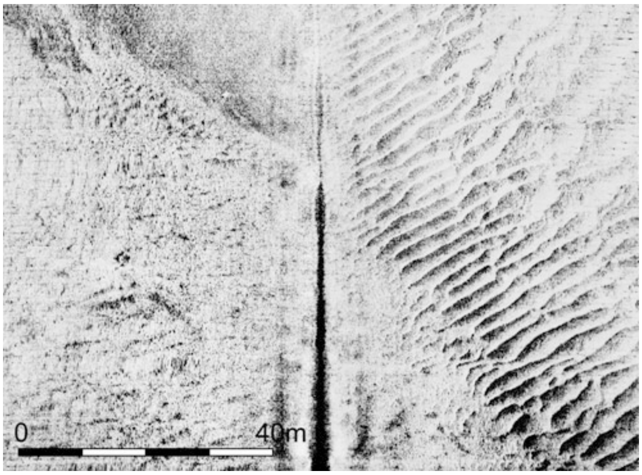
Side-scan sonar imaging allows the study of the lithological, bathymetric, and morphological characteristics of the estuarine floor. It also allows the determination of the geometry, distribution, dimensions, and orientation of the fields of bedforms and facilitates the analysis and characterization of the flow regime. Analysis of the temporal variations of these characters can significantly increase the knowledge base on sediment dynamics in estuaries.

Estuarine bedforms and tidal flow regime

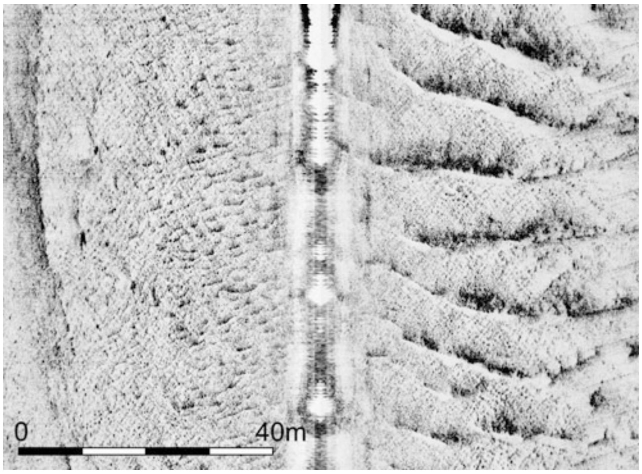
Bedforms are morphological features of sandy or gravelly beds that are normally rhythmical and that can range in size from a few centimeters to more than 15 m. Estuarine beds exhibit abundant bedforms with different morphologies and sizes dependent on the type of estuary, tidal flow regime, dimensions of the estuarine channels, and the location of the study area with respect to the longitudinal zones of the estuary. The smallest of these bedforms are



Side-Scan Sonar Imaging of Sediment Bedload, Figure 2 Principles of side-scan sonar functioning.

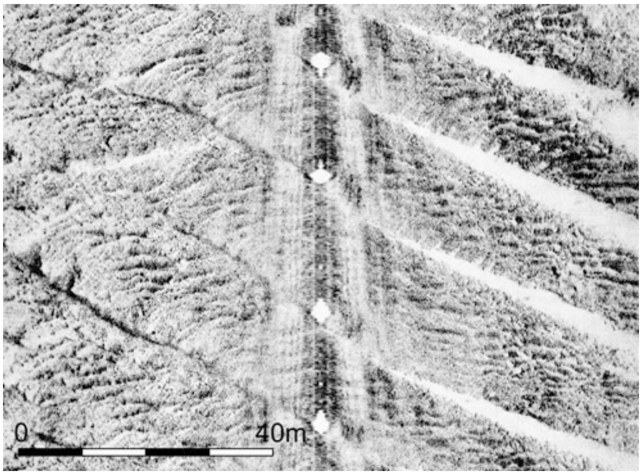


Side-Scan Sonar Imaging of Sediment Bedload, Figure 3 Transition from a plane bed located in the channel margin (*left*) to sinuous-crested megaripples (*right*). This is an example of the transition from low flow regime in the channel margins to medium flow regime in the deeper part of the estuarine sand bars.



Side-Scan Sonar Imaging of Sediment Bedload, Figure 4 Transition from sinuous megaripples (*left*) to medium sandwaves (*right*). Dark patterns located between different crests correspond with mud accumulations developed in the bedform runnels. The flow regime increases from the shallow bars to the deeper part of the estuarine channel. Mud is deposited during slack tide.

ripples; the medium-scale bedforms are megaripples or dunes; the largest metric-scale forms are sandwaves. There is a demonstrated relationship between the river/stream regime and the type, dimensions, and orientation of the dominant bedforms (e.g., Ashley, 1990; Harbor, 1998; Carling et al., 2000). In estuaries, the bedforms clearly correlate with the tidal current velocity and the time of action of ebb and flood. As a response to the reversing tidal current, the sense of asymmetrical forms can also reverse.



Side-Scan Sonar Imaging of Sediment Bedload, Figure 5 Large sandwaves with straight crests showing superimposed metric-scale sinuous megaripples. During spring tides, large sandwaves are developed in the deep estuarine channels. During neap tides, only small bedforms can migrate, covering the larger sandwaves.

Subaqueous dunes have been described by Duck et al. (2001), Van Lancker et al. (2004), and Morales et al. (2006). Similar estuarine bedforms have been described in North American estuaries by Fenster et al. (1990), Sherwood and Creager (1990), and Woodruff et al. (2001).

Examples of tidal estuarine bedforms with interpretation of the flow regime and bedload transport are presented in Figures 3, 4, and 5.

Summary

Side-scan sonar is an acoustic technique to obtain accurate images of underwater beds. In estuarine channels these images reveal a variety of bedforms that enable determination of the dominant flow regime and hydrodynamic processes, especially related to bedload transport.

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Cross-references

Estuarine Sediment Composition
Mass Physical Sediment Properties
Sediment Transport
Sedimentary Structures

SOFT COMPUTING

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Synonyms

Artificial intelligence; Computational intelligence; Evolutionary algorithms; Fuzzy logic; Machine learning; Neural networks

Definition

Soft computing refers to those computational methods that are inherently soft toward inexactness of data as well as that of problem specification. It is defined as the technique that targets the tolerance of the real world to uncertainty, imprecision, inaccuracy, and partial truths in data as well as problem specification in order to arrive at meaningful solutions (Zadeh, 1994).

Soft Computing

Following a series of works by Professor Lotfi Zadeh since the mid-1960s, usage of the term “soft computing” became prominent in the early 1990s (Zadeh, 1994). The main tools of soft computing are neural networks, fuzzy logic, evolutionary algorithms (genetic algorithms, differential evolution), and probability embedded fuzzy tools. Support vector machines, meta-heuristic and swarm intelligence, and colony optimization, particle swarm optimizations, and chaos theory also fall under the purview of soft computing (Karray and de Silva, 2004). Apart from the use of a single tool like neural network or fuzzy logic, their combinations such as neuro-fuzzy inference systems have also been beneficial (Azmathullah et al., 2008). Traditional logic recognizes only two crisp values (e.g., true or false, yes or no) and accordingly assigns the value of 0 or 1 to the variables in binary descriptions. On the contrary, the multivalued fuzzy logic assigns a range of values in between (0, 1) to the variables depending on the uncertainty and can thus account for imprecision and partial truths in data and system specification. Subjectivity can thus be replaced by objectivity in computations (Ross, 2004).

Applications of soft computing tools in estuarine and coastal fields have targeted function approximation, optimization, control, system modeling, and pattern recognition. A large number of applications pertain to the use of neural networks, fuzzy logic, and genetic algorithms to estimate and forecast waves, tides, currents, storm surge, sediment transport, salinity intrusion, foundation scour, structural optimization, control, and effects (Jain and Deo, 2006; Deo, 2010).

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Cross-references

Sediment Transport
Storm Surges
Tides

SOFT SEDIMENT COMMUNITIES

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Synonyms

Soft bottom communities

Definition

Estuarine soft sediment communities consist of groups of benthic species that frequently co-occur and show interdependence.

Description

Soft sediments – mud, silt, and sand – cover a large portion of the world's estuarine benthic environments and provide habitat for a multitude of benthic invertebrate species. Soft sediments are three-dimensional environments inhabited by many benthic species. It is well documented that soft sediment communities are not only influenced by environmental factors (e.g., sediment size, pH, organic content) but also by biotic factors (e.g., bioturbation) (see the pioneer work of Rhoads and Young, 1970). Bioturbation is linked to deposit feeding, sediment reworking, construction of burrows and tubes, and irrigation, with important effects on biogeochemical processes in surface sediments and at the sediment-water interface and on redox conditions in deeper sediments (Rosenberg, 2001).

Estuaries provide essential ecological functions (e.g., nutrient cycling), and soft sediment communities are integral to these functions (Levin et al., 2001). A large fraction of primary consumers in an estuary are found in the bottom sediments (McLusky and Elliott, 2004). The macrofauna of estuarine soft sediment communities are dominated by crustaceans, mollusks, and annelids, and there are many studies showing that the structure of these communities is an essential tool for assessing impacts on estuaries.

While the meaning of the term "soft sediment" is straightforward, the meaning of the term "community" has been a matter of debate for decades. This debate involves disagreement about populations versus communities as the appropriate level of organization to study (see Underwood, 1986 and references therein). In the estuarine ecological literature, the terms "community" and "assemblage" are often used synonymously. However, the term "community" not only refers to groups of species that co-occur in a similar habitat but also to the interdependence between those species. In contrast, the term "assemblage" refers to a collection of co-occurring species that does not require links to a specific habitat, and it does not imply interdependency (Gray and Elliott, 2009).

Summary

In the estuarine ecological literature, the terms "community" and "assemblage" are often used synonymously, but community not only refers to groups of species that co-occur in a similar habitat but also to the interdependence between those species. In the absence of experimental evidence of interdependence (or interaction), these groups should be called assemblages. Soft sediment communities are affected by the physical, chemical, and biological characteristics of the benthic habitat, and they are frequently used to assess human impacts on estuaries. Many primary consumers in estuaries are part of benthic communities that provide essential ecological functions to the system.

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Cross-references

Benthic Ecology
Biogenic Sedimentary Structures
Infauna

SOLDIER CRABS (MICTYRIDAE)

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Definition

The mictyrid soldier crab is easily identified by a number of distinctive characteristics including its relatively small body size (marble to golf ball diameter); blue coloring; prominent black eyes on short stalks; round, spider-like appearance; chelipeds (front claws) orientated in the vertical plane; and forward-walking locomotion style. McNeill (1926) provides a more detailed taxonomic description of the genus.

Introduction

Soldier crabs are semiterrestrial, intertidal crustaceans well known for their tendency to form spectacular wandering droves or swarms of hundreds to thousands of brightly colored (mainly blue) individual crabs on the tidal flat at low tide. There are eight recorded species of mictyrid soldier crabs (Table 1), and their biogeographic range incorporates coastal areas in the Indo-west Pacific region from the southern islands of Japan, through the China Sea and the Southeast Asian region, to Australia (Figure 1). Climate zones inhabited by the different species range from tropical humid, to tropical arid, to subtropical and temperate zones. Tidal ranges are also diverse with soldier crabs occurring in extreme macrotidal regimes of up to 8 m in tropical areas in northern King Sound, Western Australia, to microtidal 2 m tides in higher latitudes such as Hobart, Tasmania. Most soldier crab habitats occur in areas with semidiurnal or mixed tidal cycles and less commonly with diurnal tides (Unno and Semeniuk, 2009).

Soldier crabs are common inhabitants of estuaries, and an understanding of the habitat requirements, behavior, and life cycle of the crab combined with the knowledge of the types of habitats occurring in estuaries explains why this situation occurs. Soldier crabs are important in estuaries as diet items for shore birds and fish, and as burrowing organisms for their major bioturbation effect on substrates in terms of sediment turnover, benthic metabolism, and chemical properties (such as oxygenation and nitrogen fluxes), and sediment structuring.

Habitat requirements of the soldier crab

The highly visible emergent phase of the soldier crab as swarms or "armies" on the tidal flat surface is only a short-term expression of what is largely a cryptic, benthic existence for the crab, as infauna within the substrate during part of the low tidal and all of the high tidal periods. Soldier crabs have an intimate relationship with their habitat as they reside beneath the surface, living, feeding, and moving around in their subsurface habitat. Generally, they do not occur deeper than 30 cm unless pursued by predators. The sediment characteristics of their habitat are mostly fine to medium sand (quartz or calcareous) with a small percentage of interstitial mud and organic matter. *Mictyris longicarpus*, the largest of the soldier crab species, has been found in a more diverse range of sediment types from coarse or very coarse sand to muddy sand. Sediment pellicular water and groundwater salinity are *circa* marine salinity, but soldier crabs are osmoregulators and can tolerate considerable variations in salinity (Barnes, 1967). With capillary tube lungs as well as gills (Maitland and Maitland, 1992), soldier crabs are obligate air-breathers, not subtidal crabs, and therefore will drown if the frequency of inundation is too high. However, they do require a moist environment, and a shallow water table is necessary at low tide to allow for pellicular water to be present in the substrate via capillary rise. However, the substrate must be moist enough for the crabs to be able to pelletize the sand. Similar to other intertidal crabs, such as *Dotilla* or *Scopimera*, soldier crabs are filter feeders, scooping sand into their water-filled buccal cavities where the organic material is filtered off into the gut and the heavier sand particles separated and compacted into a discard pellet (Quinn, 1986).

Unlike many other intertidal crabs such as *Uca* and *Sesarma*, soldier crabs do not have permanent burrows but create a subsurface circular air cavity by burrowing into the substrate from the surface in a cork-screw motion, sealing off the top of the hole with sand pellets in a characteristic rosette pattern. The crabs move through the subsurface of the sediment, dragging the air bubble with them by pushing sand from one side of the bubble to the other. Crabs will reside below the surface and carry on feeding in the air bubble during high tide when the substrate is saturated (Unno and Semeniuk, 2008). After surface exposure at low tide, if the crabs have exhausted or lost their air bubble, they will emerge briefly and then create a reentry rosette while renewing their air cavity. It is the presence of the mobile air cavity that constrains the soldier crab to remain relatively close to the substrate surface, and as a result, the most important habitat requirement for soldier crabs, and the one that determines where many soldier crab populations reside within estuaries, is that of wave energy. Waves influence the stability of the substrate, with high energy waves creating megaripples and massive shoal movements that would disrupt soldier crab air cavities, eliminating them from the area. Lower wave energy can result in an increase in the mud content of substrates to the extent that soldier crabs are eliminated from the environment as they can no longer feed or create air cavities. The balance for the soldier crab habitat is one between the appropriate level of wave energy and mud influx (Unno and Semeniuk, 2009).

Soldier crab behavior and life cycle

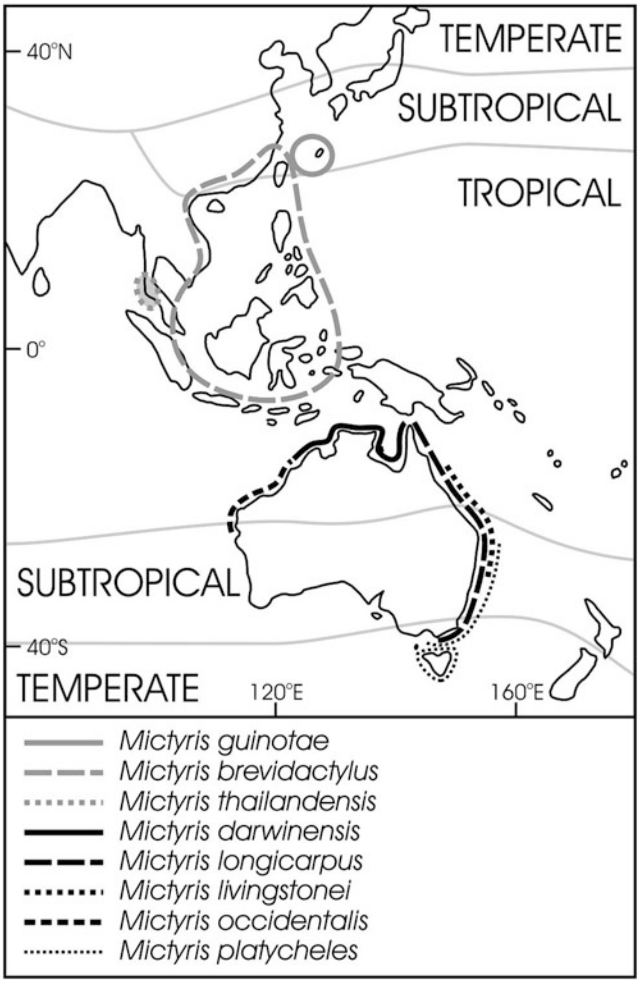
Once a soldier crab population has colonized a suitable habitat, the complexity of soldier crab behavior and the concomitant organismal structures (ichnos) created by the crabs become evident. The complexity of behavior and the resultant ichnological structures increases with the age of the crab. After a planktonic larval stage, soldier crabs settle as juvenile recruits into the substrate and their presence is evinced by small clots of sand on the surface at low tide. As the crabs molt and grow, juvenile crabs can be seen emerging after exposure at low tide and creating small crater-like structures. Adult crabs have the greatest diversity of ichnological products with their emergent phase and feeding activities on the surface resulting in short vertical subsurface shafts, exit holes, scrape marks, discard feeding pellets ("pseudofecal" pellets), and reentry rosettes (Figure 2). Discard pellets can densely cover the tidal flat for tens to hundreds of square meters. Infaunal crabs that remain in the substrate at low tide but are feeding close to the surface are indicated by individual circular

Soldier Crabs (Mictyridae), Table 1 Recorded species of *Mictyris* Latreille, 1806, and their biogeographic range and images

Species	Biogeographic range	General features	Image
Mictyris longicarpus Latreille, 1806	Australia: Cape York, Queensland to Wilson's Promontory, Victoria	Typical size: 30 mm Eyes: large Color: blue with white sides (branchial regions); cream legs with narrow red bands at base and middle joints	
Mictyris brevidactylus Stimpson, 1858	Taiwan, Kinmen, China, Hainan Island, Vietnam, Hong Kong, Southeast Asia, Singapore, Indonesia, Karakelong Island, Ambon Island, Bawean Island	Typical size: 20 mm Eyes: medium Color: blue, paler blue sides; cream legs with broad red bands at base of walking legs (Photo: Hsi-Te Shih)	
Mictyris platycheles H. Milne Edwards, 1852	Australia: Moreton Bay, Queensland, northern New South Wales, to Tasmania	Typical size: 20 mm Eyes: medium Color: dark blue with reddish-purple sides; reddish-cream legs	
Mictyris livingstonei McNeill, 1926	Australia: Cookstown, Queensland to Trial Bay, New South Wales	Typical size: 15 mm Eyes: small Color: blue with lighter sides; cream legs	
Mictyris occidentalis Unno, 2008	Australia: northern King Sound to Shark Bay, Western Australia	Typical size: 15 mm Eyes: medium Color: blue with paler blue or pinkish sides; cream or orange legs	
Mictyris guinotae Davie, Shih & Chan, 2010	Japan: Ryukyu Islands (Nansei Shoto)	Typical size: 15 mm Eyes: medium Color: blue, with paler blue or brownish sides; cream legs (Photo: Shawn Miller)	
Mictyris darwinensis Unno and Semeniuk, 2011	Australia: northern King Sound, Western Australia to Cape York, Queensland	Typical size: 15 mm Eyes: medium Color: slaty blue, with pale reddish or brown sides; cream legs	

Soldier Crabs (Mictyridae), Table 1 (Continued)

Species	Biogeographic range	General features	Image
Mictyris thailandensis Davie, Wisespongpand & Shih, 2013	West coast of Thailand: Andaman coast between Ranong Province and Pakbara Beach, Satun Province	Typical size: 15 mm Eyes: medium Color: light blue, with pale blue to cream sides; white chelipeds and fawn walking legs (Photo: Puntip Wisespongpand)	



Soldier Crabs (Mictyridae), Figure 1 Map of coastal central Asia and Southeast Asia and Australia showing the distribution of the eight species of *Mictyris*.

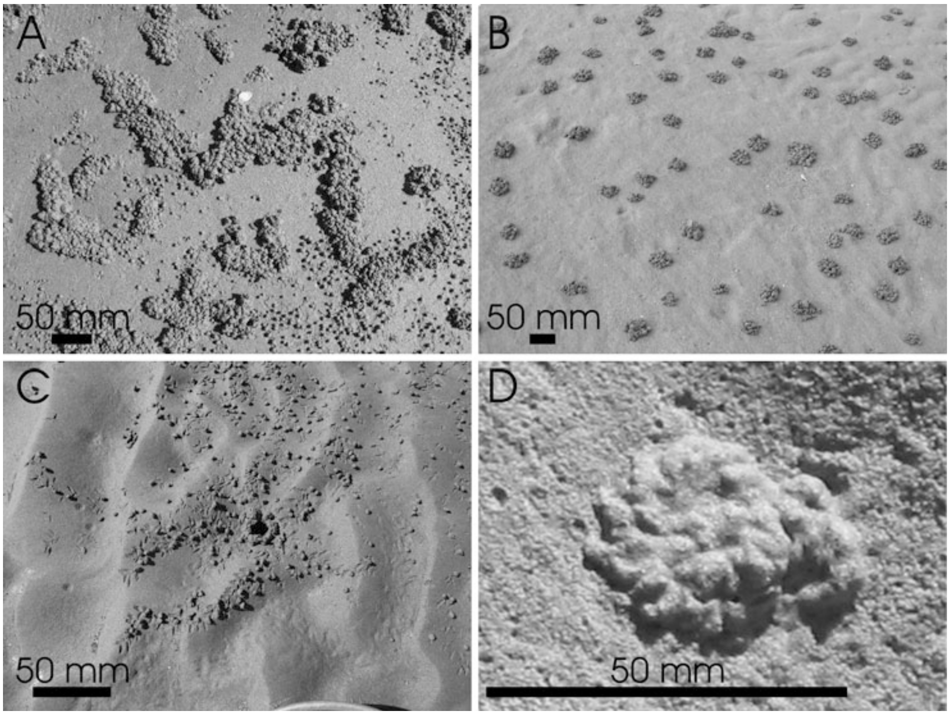
pustular structures and elongate pellet-roofed tunnel workings appearing on the tidal flat surface. Dense tunnel workings may coalesce to form extensive mats of pellet-roofed tunnels over the tidal flat resembling "pustular mats."

Soldier crabs function ecologically as major bioturbators and aerators of sediment in the intertidal ecosystem and as food items both for avifauna at low tide and for nekton on the high tide. Their daily ingestion of the substrate and cycling of groundwater in conjunction with a physiological tendency to accumulate heavy metals suggests that they can act as bioindicators of pollution events. They have an effect on meiofauna assemblages and nutrient fluxes within the substrate over an area that may be kilometers in extent, depending on the scale of the habitat.

Soldier crabs recorded in estuaries

Soldier crabs have been documented in estuaries throughout their Indo-west Pacific biogeographic range and studied in regard to various aspects of their ecology including activity patterns, physiology, burrowing behavior, swarming behavior, feeding behavior, reproductive behavior, population dynamics, benthic metabolism, and the effect of soldier crab feeding on sediment and meiofauna (Cameron, 1966; Kelemec 1979; Farrelly and Greenaway, 1987; Maitland and Maitland, 1992; Dittman, 1993; Shih, 1995; Dittman, 1998; Rossi and Chapman, 2003; Sadao, 2003; Webb and Eyre, 2004; Takeda, 2005). The importance of soldier crabs in estuaries has been recognized for their role as burrowing organisms in rapidly reworking sandy substrates and effecting oxygenation, nitrogen fluxes, and other chemical changes. They effect sediment turnover and sediment structuring to a depth generally of 10–15 cm (Unno and Semeniuk, 2008), though they can occur to a depth of 30 cm.

River mouths are often locations for major towns or cities, and consequently researchers are located close to their study sites. In Japan and Southeast Asia, mountainous terrain and monsoonal climate result in short rivers draining to the coast and developing estuaries. Most estuaries are of the ria, coastal plain (flooded valley), or deltaic estuarine types and are bordered by dense stands of mangroves (Fairbridge, 1980). For instance, *M. guinotae* has been studied on shore-parallel platform tidal flats adjacent to mangroves in the Nadasa River estuary on Iriomote-jima, Japan, and, similarly, Shih (1995) observed *M. brevidactylus* on sandy tidal flats bordering mangrove woodlands in the Tanshui River estuary in Taiwan.



Soldier Crabs (Mictyridae), Figure 2 Ichnological products of *Mictyris*. (**a**) Meandering and linear pustule structures of Unno and Semeniuk (2008), which are pellet-roofed, shallow, horizontal tunnels. (**b**) Pustules produced by crabs re-excavating their air bubble. (**c**) Exit hole, feeding scrape marks, and discard feeding pellets ("pseudofecal" pellets). (**d**) Close-up of a reentry rosette, comprising a central plug and an outer ring that has a vague curved, radial structure, formed by crab burrowing for reentry into the sediment.

For the Australian species, a multitude of rivers draining eastwards from the Great Dividing Range and Blue Mountains provide many estuarine habitats for soldier crab species along the eastern coastline. *M. longicarpus* has been well studied in the Brisbane River and Pine River estuaries draining into Moreton Bay, Queensland (Cameron, 1966; Dittmann, 1998) and inhabits many estuarine environments from northern Queensland to southern Victoria (Figure 1). *M. livingstonei* occurs from northern Queensland to the Macleay River estuarine complex in Trial Bay, New South Wales. *M. platycheles* occurs along on the central and southern mainland Australian coast, as well as in numerous estuaries along the northern, eastern, and southern coastlines of Tasmania (Figure 1). In the Northern Territory and northern Western Australia, drainage from the highlands and tectonic blocks result in extensive river outflow to the coast forming estuaries wherein *M. darwinensis* resides (e.g., Keep River estuary). *M. occidentalis* inhabiting the northwest Western Australian coastline is an exception to the soldier crab trend towards dominantly estuarine inhabitation. Within the crab's biogeographic range, the climate is arid to semiarid resulting in the development of few large rivers and estuaries. Rivers are dominated by marine salinities and only function intermittently as estuaries every 5–10 years during times of flood; an example is the Gascoyne River estuary where soldier crab populations reside on deltaic (strand plain) shoals and upstream on sandy creek banks and point bar shoals.

Range of habitats available in estuaries

Soldier crabs inhabit a wide variety of large-scale coastal settings, including estuaries, barred lagoons, tidal creeks, delta strand plains, tidal flats, beach/dune shores, and limestone barrier coasts. An estuary by definition is the area where freshwater from rivers intermix with marine water on a seasonal or intermittent basis resulting in a variation of salinity ranging from marine to brackish to fresh depending on the fluvial input. Factors influencing the development of local habitats for soldier crabs within an estuary include the large-scale geomorphology of the coast, coastal and marine processes, and fluvial hydrology and sediment supply.

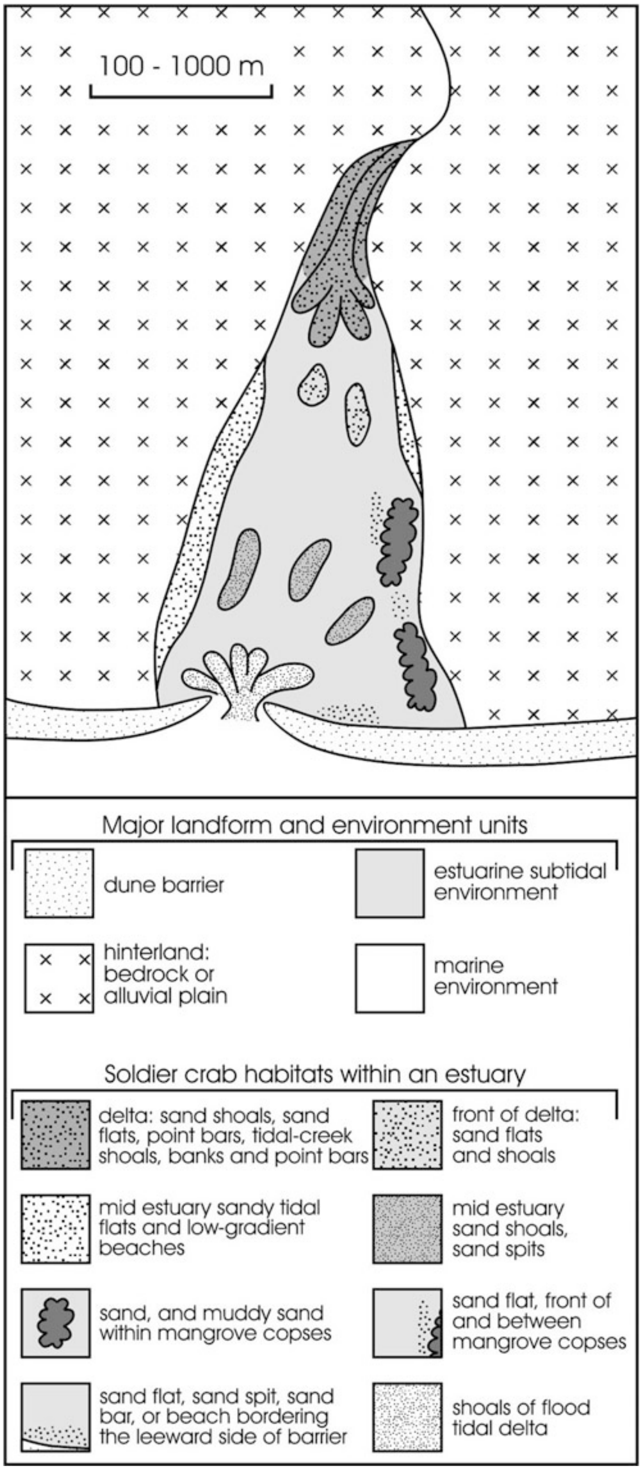
The presence of soldier crabs in an estuary depends on climate setting and coastal setting of the estuary, the type of estuary, as well the sedimentological and hydrological regime prevalent in the estuary. An estuary may have many local habitats suitable for soldier crabs but can be situated in a region that due to remoteness from existing soldier crab populations, or lack of regional currents to deliver larval recruits, is depauperate of mictyrid crabs. The Murchison River estuary in Western Australia is such an example – despite having suitable local sand shoals, it is isolated by the Zuytdorp Cliffs and rocky coastline from the nearest soldier crab population 200 km to the north at Shark Bay (Unno and Semeniuk 2009). Of the various types of estuaries, the ria or flooded river valley and the bar-and-lagoon types (with an inlet opening to the ocean) are most likely to host soldier crab populations as they provide sheltered, low-wave energy environments.

There is a salinity gradient within an estuary from freshwater to brackish water near, at, or within the river mouth to brackish water to marine water in the central estuarine reaches, to marine (and seasonally brackish water) at or near the estuary inlet. Soldier crabs inhabit estuaries from the marine saline parts to the brackish water parts, occupying tidally exposed sandy substrates between the Mean Low Water Neap (MLWN) and Mean High Water Spring (MHWS) tidal levels.

With respect to sandy substrates, at the local scale, soldier crabs inhabit a wide range of sheltered (low-wave energy) stable sandy habitats (Figure 3), including sand bars, recurved spits, shoals, and depressions leeward of spits; low gradient sloping beaches; shore-parallel shoals; sheltered areas of sand on rock pavements; sheltered areas among rock on sandy beaches; low-mid-high tidal sand flats; tidal flats on shore-parallel sand platforms and sandy estuarine river channel banks; mid-channel sand shoals, mid-bay or mid-estuarine sand shoals; banks, shoals, and point bars of tidal creeks; shoals, sand flats, and point bars of intra-estuarine deltas; localized sand flats shoreward or seaward of mangroves; sandy to muddy sand intertidal platforms adjacent to and within mangrove woodlands; sand flats among mangroves; and flood-tidal delta shoals (Unno and Semeniuk, 2009).

Summary

Soldier crabs are common intertidal crabs inhabiting suitable sandy habitats within estuaries in the Indo-west Pacific region. The sheltered low-wave energy environments present in estuaries include habitats such as sand bars, recurved spits, aligned sandy beaches, tidal flats on shore-parallel sand platforms, sandy estuarine river channel banks, mid-channel sand shoals, mid-bay sand shoals, intra-estuarine delta and tidal delta shoals, and sandy to muddy sand intertidal platforms adjacent to mangrove woodlands. The fluctuating water salinities within estuarine environments are tolerated by soldier crabs through osmoregulatory mechanisms. While notable for their habit of swarming in large "armies" of small blue crabs on the tidal flat at low tide, they largely live infaunally in round air cavities within the substrate, relatively close to the surface. Development of the crab from juvenile to adult is accompanied by increasingly complex behavior and corresponding complexity in ichnological products and emergent behavior.



Soldier Crabs (Mictyridae), Figure 3 Idealized map of an estuary showing a selected range of habitats that *Mictyris* inhabits. The salinity of their seaward habitats is marine (and may be seasonally brackish); the salinity of their near-riverine and deltaic habitats is brackish (and may be seasonally marine).

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Cross-references

Blue Crabs
Fiddler Crabs

SPECIES RICHNESS

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Synonyms

Number of species per area/region/ecosystem

Definition

Species richness is the simplest way to describe biotic community and regional diversity (Maguran, 1988). It refers to the number of species in an area, biotic community, or ecosystem. Species richness does not take into account the abundances of the species or their relative abundance distribution, but rather the number of species in a particular area considering their phylogenetic differences as part of the diversity (Smith and Smith 2001). Different species concepts have been used in species richness studies (i.e., biological, ecological, evolutionary, and phylogenetic species concepts). An unified species concept has recently been proposed by de Queiroz (2007).

Description

The observed species richness is affected by the general area of sampling, heterogeneity of the habitat, trophic structure of the area, geographic region, and season of sampling. The species richness can vary considerably in different habitats, seasons, and geographic regions. The addition of new species with increasing sampling effort can be shown by a species accumulation curve (Bower and Zar, 1995). Increasing the area sampled can also increase the observed species richness both because large areas are environmentally more heterogeneous than small areas and because more individuals may inhabit these areas. Species richness is a fundamental measurement of community and regional diversity, and it underlies many ecological models and conservation strategies (Gotelli and Colwell, 2001).

With species richness studies, ecologists often employ rank abundance curves, which are graphs ranking the most abundant species to the least abundant (Adams, 2009). They can be shown as a plot of number of species vs. the number of individuals on a logarithmic scale that usually yields a normal distribution. This is because environments are usually undersampled, especially in high diversity systems or regions. Singlets make up the middle peak of the distribution, and the more sampling that is conducted, the more the curve will shift to the right. The number of unsampled species in cases of undersampling can be roughly estimated using the Chao estimator, in which $S_{estimate} = S_{observed} + F_1^2/2F_2$ where F_1 is the number of singletons sampled and F_2 is the number of doublets. There is also a method of estimating what percent of the total species is represented in a sample, called Good's coverage estimator, in which Coverage = 1 − (the number of individuals in species/total number of individuals). These estimator equations allow researchers to determine how their limited sampling relates to the entire sampled population (Chao, 2005).

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Cross-references

Shannon-Weaver Diversity Index

SPECIES ZONATION

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Definition

Species zonation is the occurrence of species or groups of species in distinct bands or zones coincident with or related to environmental gradients.

Introduction

Species zonation is the occurrence of species or groups of species in distinct bands or zones coincident with or related to environmental gradients. It is a common feature observed within estuaries where the distribution and abundance of its biota respond to the combined and interactive effects of steep environmental gradients and food availability and to interspecies competition, herbivory, and predation (Paine, 1974; Pennings and Callaway, 1992; Levinton, 1995). The distribution and abundance of individual species (as a result of their tolerances to the combined effects of the environmental gradients) results in compositional changes in biotic assemblages across these gradients. For vegetation, species zonation is often manifest as visually distinct vegetation communities; for rock-inhabiting communities, it is commonly manifest as visually distinct bands of different encrusting shelly organisms, epibenthos, and algae. For infauna, it is manifest as compositional and abundance changes in the communities across the habitat as determined by sampling and mapping. In areas of less steep environmental gradients, species often occur in less differentiated formations, such as mottled mosaics or with diffuse zonation.

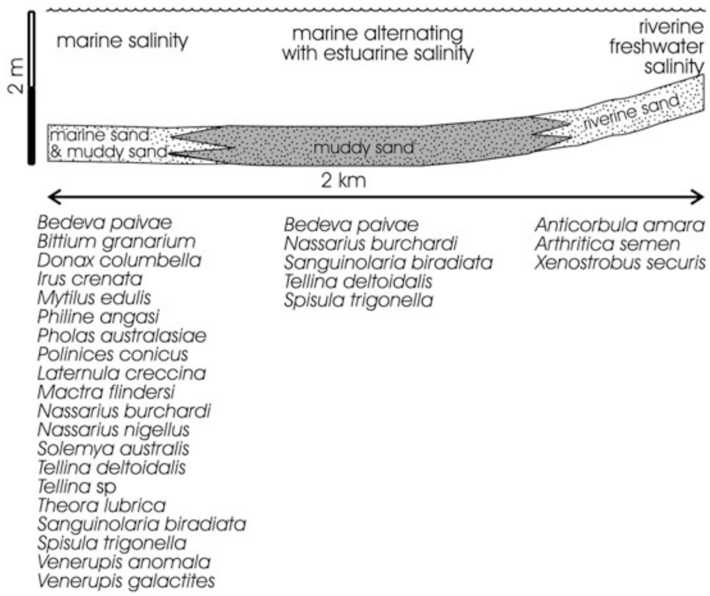
The description and discussion of species zonation provided here is for a positive estuary that grades from freshwater at the river entrance to marine at the estuary mouth, but the principles of species response also apply to inverse estuaries where evaporation from the surface water exceeds the freshwater runoff entering the estuary.

Variation on zonation in different environmental situations

In estuaries where there is a strong seasonal to perennial delivery of freshwater via subterranean seepage from supratidal environments (Semeniuk, 1983; Cresswell, 2000), the gradient of increasing salinity and pore-water content upslope can be locally reversed at the contact of the tidal flat with the supratidal zone. Where frequency of inundation and concomitant change in wave energy and tidal current energy also result in a differentiation/partitioning of sediment grain sizes across the tidal flat (from sand in low tidal areas to mud in high tidal areas), there may be three gradients across the tidal flat, that of salinity, pore-water content, and sediment grain size. As a consequence, biota responding to these gradients in concert often forms distinct zonation across the flat.

Heterogeneous distribution of habitats and lack of species zonation

In many estuaries there are mosaics of habitats and mosaics of biotic assemblages and occurrences of species relating to these habitats. Due to the particular and determinative environmental conditions present in a habitat, the species composition of biotic assemblages can vary markedly between habitats. Estuaries with heterogenous distribution of habitats can result in a mosaic of biotic assemblages corresponding to these habitats within the



Species Zonation, Figure 1 Zonation of benthic mollusks from river to sea across an estuary showing response to an environmental gradient of open-water salinity (information from Semeniuk and Wurm, 2000). The change in mollusk composition is from species-depauperate, fluvial-dominated assemblages to species-rich, marine-dominated assemblages.

estuary. The distribution of assemblages across and along the estuary in these cases is not zoned; e.g., an estuary comprised of deepwater mud basins, shallow-water subtidal shore-parallel sandy platforms, tidal flats, and hummocky tidally exposed shoals may have distinctive fauna and flora in each of these habitats, but a transect across the estuary will not illustrate species zonation nor assemblage zonation but rather habitat-specific occurrences of assemblages. In contrast, species zonation, if present in such estuaries, may be evident within a given habitat that has environmental gradients *within* the habitat.

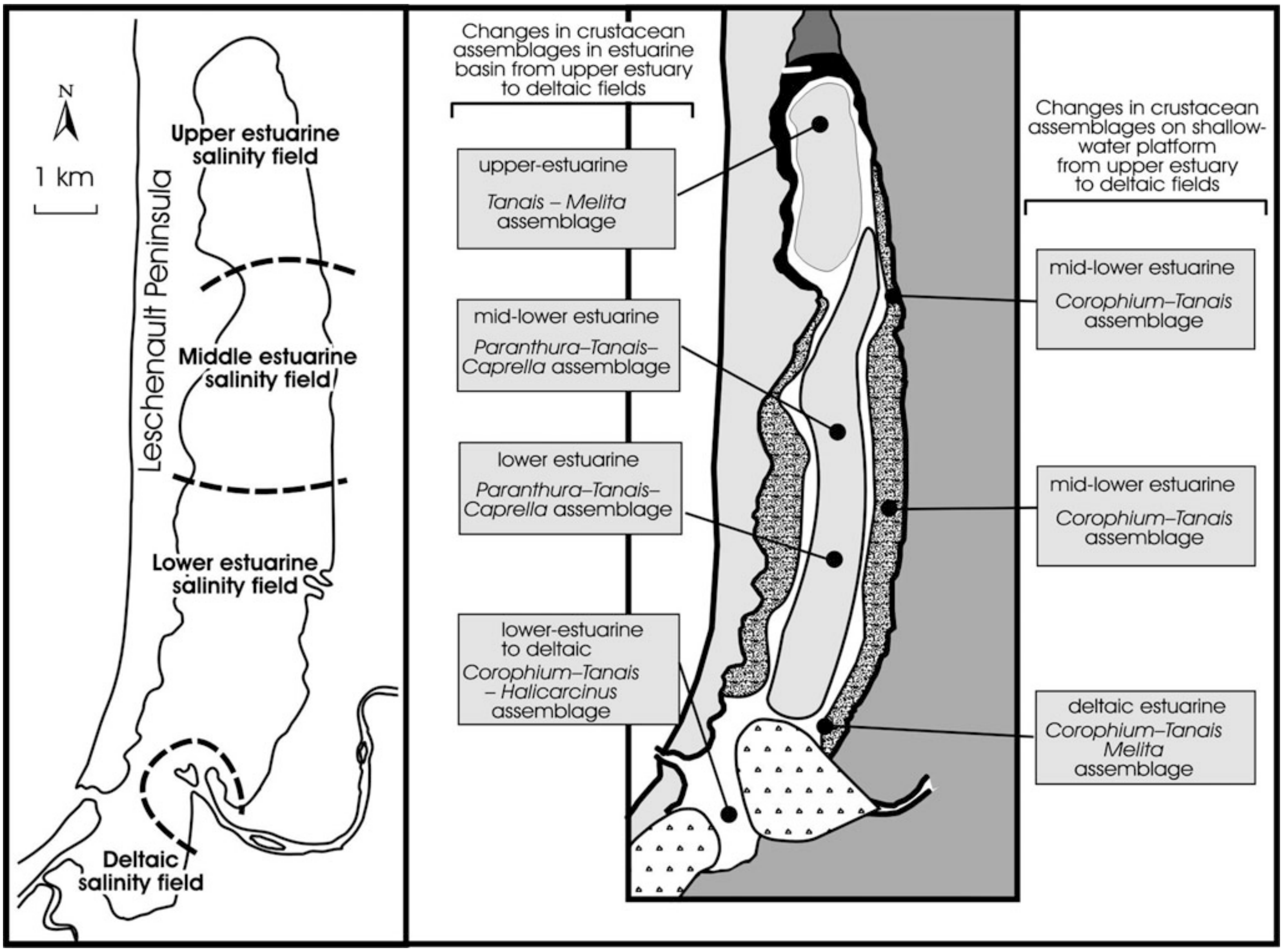
Some environmental determinants underpinning zonation

The main environmental determinants for forcing species zonation that are examined in this work are pore-water salinity, open-water salinity, substrate type, and inundation. Each of these factors is discussed below in relation to effects on species zonation. In the real world, it is the combination of each of these physical environmental gradients, plus the interaction between species, that determines the final zonation of species. Other environmental factors and gradients also influence zonation; these include wave energy, tidal currents, extent of water turbidity, water depth, temperature, degree of light penetration, pH, and nutrients. They influence the survivorship of a given species, determine the suitability of a habitat for an organism, and also affect the microbiota in the environment that then influence the occurrence of macrofauna and macroflora.

Saline conditions can be lethal to biota adapted for freshwater conditions, and freshwater can be lethal or debilitating to biota adapted for saline conditions. So, the salinity of open estuarine water or that of sediment pore water (influencing those biotas residing in the substrate or having their roots in the sediment) controls the occurrence and functioning of the various species in an estuary and results in their occurrence or absence and in their differential abundance. In the case of substrates, some biotas are adapted solely for inhabiting sandy substrates, and the occurrence of mud interferes with their feeding or respiratory processes while, conversely, organisms adapted to quiescent muddy conditions cannot tolerate mobile sandy substrates. In the case of inundation, how frequently an area in the tidal zone is inundated will influence the distribution of those biotas that require nearly continuous inundation for survival (Pennings et al., 2005). The gradient of inundation affects the extent that tidal zones are exposed to solar radiation, winds, evaporation, and groundwater draining (which depletes sediment moisture content), all of which influence the survivorship and population dynamics of a given species.

Open-water salinity gradient

The salinity of open water in an estuary controls species occurrence, distribution, and zonation and is expressed in a major gradient along an estuary (Figure 1). Salinity can vary from marine near the estuary entrance to fluctuating marine and brackish water and freshwater in the central estuary, to freshwater proximal to the river mouth or up-channel in a river. This salinity gradient results in

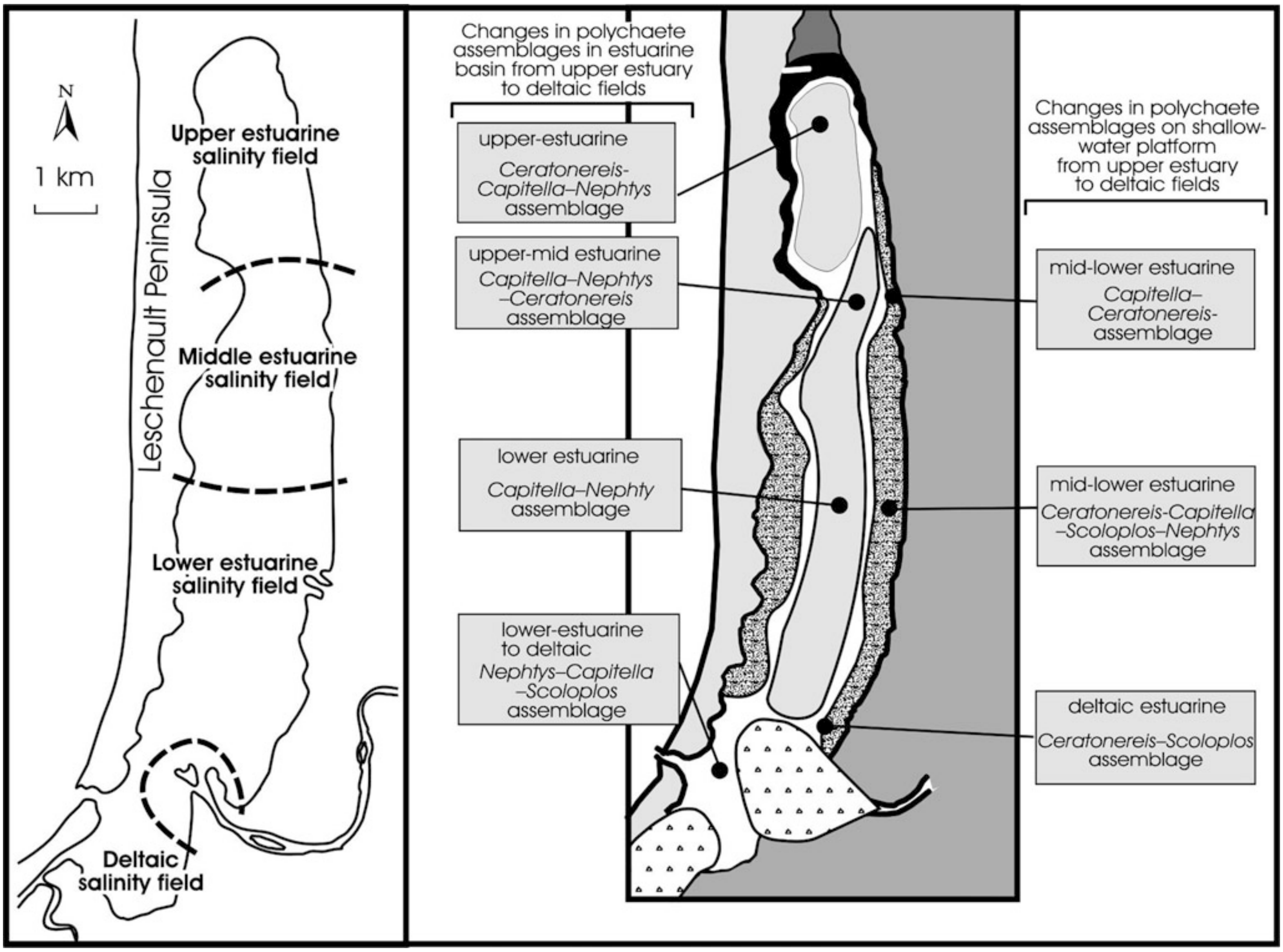


Species Zonation, Figure 2 Zonation of crustacean assemblages within habitats (a shore-parallel subtidal sand platform and a subtidal basin) crossing an environmental gradient of open-water salinity from upper estuary to lower estuary and deltaic, demonstrating the partitioning of the habitat into biotic zones (information from Semeniuk, 2000). In this case, the habitats are shore-parallel subtidal sand platforms and a basin (with similar substrates along their length) that cross three salinity fields in an elongated estuary.

major biotic compositional changes, expressed as zonation along the length of an estuary, from marine-dominated assemblages at one end to freshwater assemblages at the other, with the marine and brackish water/freshwater central environment with species that are adapted to fluctuating salinity conditions and those that can tolerate a wide range in salinity (Semeniuk and Wurm, 2000).

Pore-water salinity gradient

Pore-water can be different from open-water salinity in that it may remain stasohaline (relatively constant), while overlying open estuarine water fluctuates from marine to brackish water to freshwater. In addition, being in contact with sediment, its geochemistry, and its organic matter, pore water has a more diagnostic and variable chemical and biochemical signature than open estuarine water in terms of pH, Eh, cationic content, nutrients, and dissolved gases. Focusing on pore-water salinity, while open-water salinity may influence the occurrence and distribution of epifauna and nekton, pore-water salinity has influence on the survivorship and distribution of infaunal benthic organisms and on plants, because the fauna is directly in contact with pore waters, and plants have their roots immersed in such water and draw on pore waters for transpiration and nutrient transfer (Pennings and Callaway, 1992; Silvestri et al., 2005; Unno and Semeniuk, 2009). Pore water of higher salinity adversely affects fauna and flora not adapted to those conditions and hence can eliminate species. Conversely, pore water that is not saline enough can also physiologically affect fauna and flora adapted to more saline conditions which are then eliminated from this “fresher” zone.



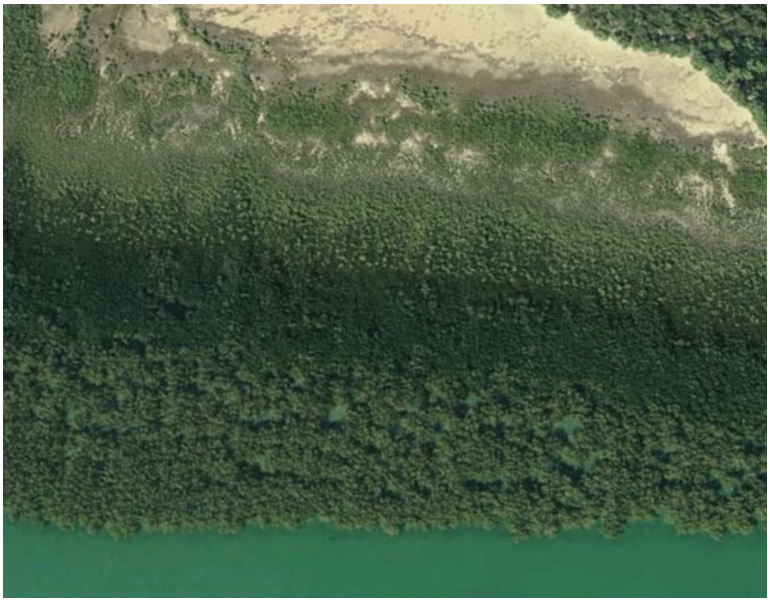
Species Zonation, Figure 3 Zonation of polychaete assemblages within habitats (a shore-parallel subtidal sand platform and a subtidal basin) crossing an environmental gradient of open-water salinity from upper estuary to lower estuary and deltaic, demonstrating the partitioning of the habitat into biotic zones (information from Dürr and Semeniuk, 2000). In this case, the habitats are shore-parallel subtidal sand platforms and a basin (with similar substrates along their length) that cross three salinity fields in an elongated estuary.

Substrate

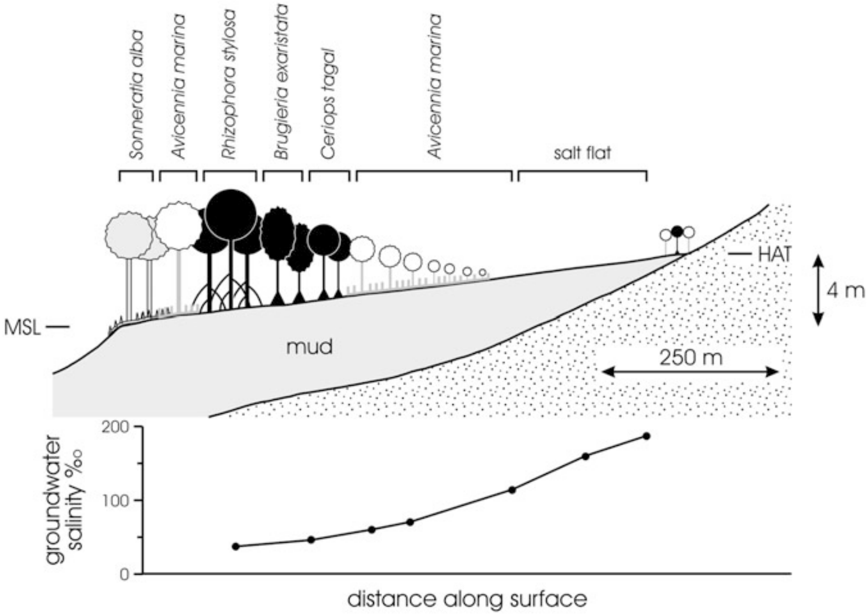
Substrate grain sizes have a major effect on biota occurrence, abundance, and functioning (McLachlan, 1996; Semeniuk and Wurm, 2000; Unno and Semeniuk, 2009). Mobile sand, for instance, agitated by current and waves and devoid of interstitial mud and organic matter, is a specific environment that only specialized fauna can inhabit. Such environments are well oxygenated and transmissive to pore waters and contain little or no organic material interstitial to the sand particles. Faunas that require a proportion of mud to line their burrow walls, or substrates that are stable (non-mobile) to build burrow structures, or feed on microbiota interstitial to sand grains find mobile substrates unsuitable. Stable sand not mobilized by currents and waves and containing some interstitial mud and organic matter provides a specific environment for specialized fauna; such environments are not well oxygenated and less transmissive to pore waters and contain organic material interstitial to the sand particles. The Western Australian soldier crab, and its associated invertebrate community, is an example of fauna adapted to inhabiting a stable-sand substrate specifically with medium/fine sand and <10 % mud (Unno and Semeniuk, 2009). On the other hand, faunas that burrow and forage through organic-rich mud and build permanent burrows find mobile sand and even slightly muddy stable sand unsuitable.

Inundation

The frequency of inundation has a large influence on the occurrence and abundance of species across tidal flats (Pennings and Callaway, 1992; Mitsch and Gosselink,



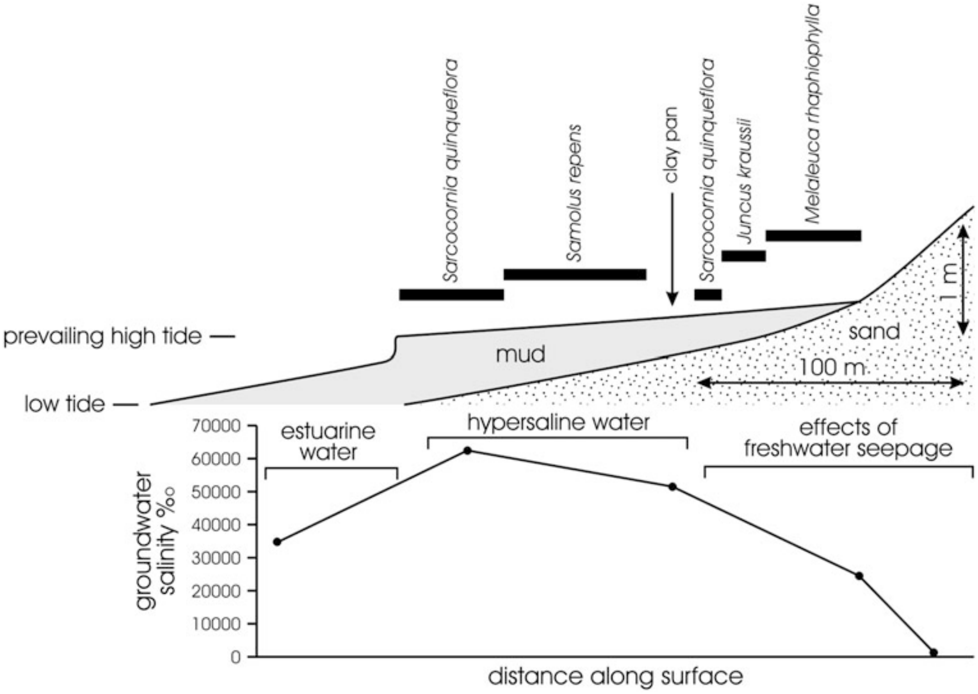
Species Zonation, Figure 4 Well-zoned mangroves on a tidal flat (a shore-parallel macrotidal intertidal mud flat in Darwin, Australia). The distinct color-evident mangrove bands from seaward with the main species identified being *Sonneratia alba* followed by *Avicennia marina*, a dark zone of *Rhizophora stylosa* followed by *Bruguiera exaristata*, then *Avicennia marina* and subordinate *Ceriops tagal*, and (most landward) *Avicennia marina* heath with samphires. The width of the mangrove band from seaward to the mangrove-free salt flat is ~340 m.



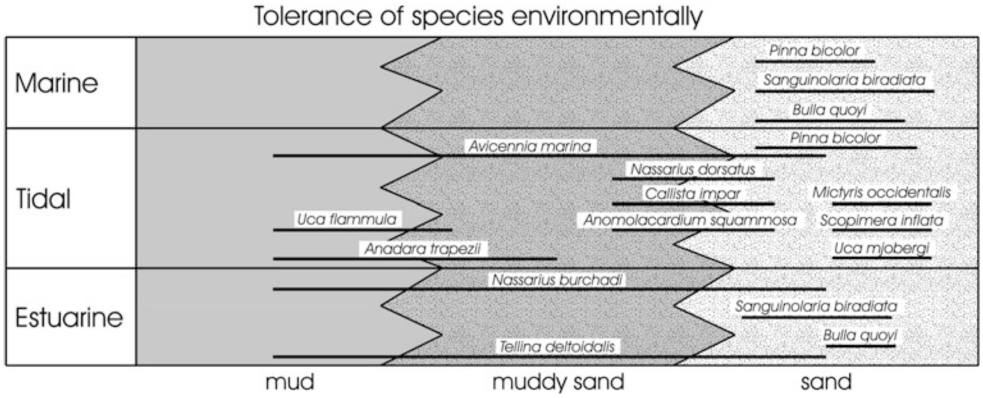
Species Zonation, Figure 5 Profile showing well-zoned mangroves on a tidal flat in the Lawley River Estuary, Western Australia, in relation to tidal levels and to groundwater salinity (cf., Semeniuk, 1983).

1993). The higher parts of a tidal flat are exposed for longer periods than low tidal parts. The gradient from low tidal flat to high tidal flat is thus subject to increasing effects of solar radiation and wind and hence evaporation and desiccation. The gradient from low tidal flat to high tidal flat is also subject to increasing internal drainage such that higher flats at the time of low tide have drier substrates than lower flats. This gradient of increased salinity and decreased pore-water content from low tidal flats to higher tidal flats results in absence and presence of species

Species Zonation, Figure 6 Well-zoned saltmarsh on a tidal flat (a shore-parallel microtidal intertidal mud flat in the Western Port, Victoria, Australia). The distinct color-evident vegetation bands with the main species identified from seaward are a seaward fringe of *Avicennia marina* with a shore-parallel band of mangrove-free patches within the mangrove zone, followed by a lighter-toned zone of fine parallel bands composed of mixed assemblages of *Sarcocornia quinqueflora* and *Samolus repens* and then a wide relatively dark "samphire" zone of *Tecticornia arbuscula* assemblage; the most landward zone is shore-parallel copses of *Melaleuca ericifolia* (cf. Bridgewater, 1975). The maximum width of the saltmarsh band in this image is ~250 m.



Species Zonation, Figure 7 Profile showing well-zoned saltmarsh on the shores of the Leschenault Inlet Estuary, Western Australia, with the main species identified in relation to tidal levels and to groundwater salinity. The landward edge of the high tidal saltmarsh receives freshwater seepage that dilutes the groundwater hypersalinity (cf. Cresswell, 2000; Pen et al., 2000).



Species Zonation, Figure 8 Simplified diagram modified from Semeniuk and McNamara (2009) to specifically illustrate the environmental tolerances of selected species in subtidal and tidal estuarine and marine environments in terms of sand, muddy sand, and mud substrates. Some species are habitat restricted, because of grain size influences, food sources, and salinity (e.g., the mollusks *Bulla quoyi* and *Sanguinolaria biradiatai*, the crabs *Mictyris occidentalis*, *Scopimera inflata*, and *Uca flammula*), while others inhabit a range of substrate types (e.g., the mollusks *Tellina deltoidalis* and *Nassarius burchardi*). Some species cross environmental boundaries, occurring in estuarine and marine settings. This diagram does not address depth occurrence or wave and tidal effects on species occurrence.

and a zonation of species across the tidal flat. For example, experimental results have shown that for salt-marsh vegetation, periodicity of inundation can be a major determinant of survival, as it determines oxygen availability, and only plants adapted to these conditions are able to tolerate such environments (Pennings and Callaway, 1992; Mitsch and Gosselink, 1993).

Broad-scale zonation

An estuary may be comprised of a sequence of habitats that intergrade from river to sea, forming a gradient of environments. In this context, biota responding to the gradient of changing environments will form gradational zones. For example, the gradient from river to estuarine basin involves tidal sand, subtidal muddy sand, and deeper water mud, with a gradient of open-water salinity of freshwater to brackish water. Here, species zonation reflects the gradient or sequence of habitat types, as well as the along-estuarine environmental gradient of salinity and substrate grain size. This is species zonation across multiple habitats (Figures 2 and 3).

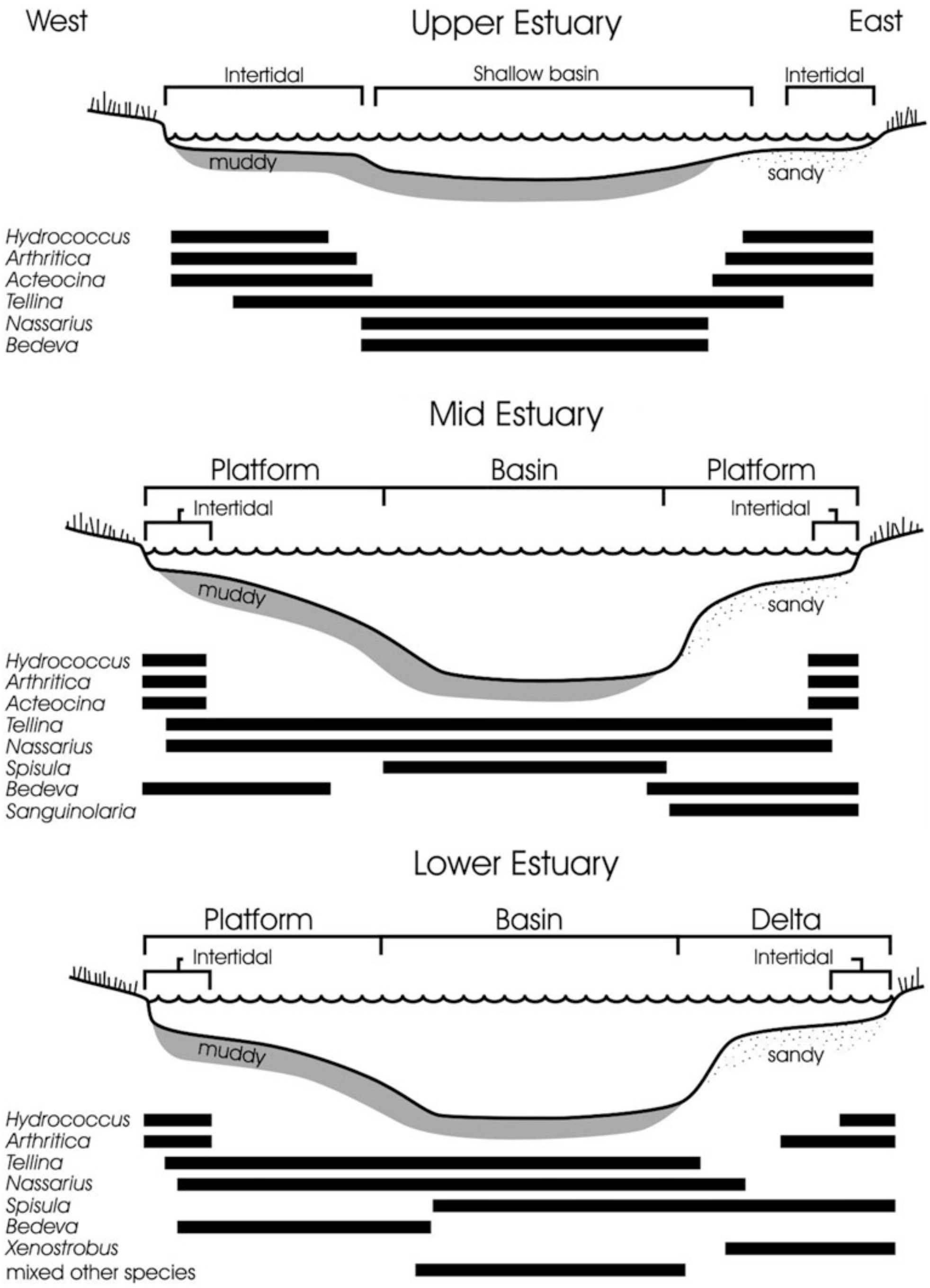
Finer-scale zonation

Zonation within a habitat occurs when there are strong and steep environmental gradients. For instance, on tidal flats where daily tidal exposure introduces inundation, evaporation, salinity, and moisture content, the local species respond to these environmental conditions and occur in distinct and discrete bands. Where there is an aggregate of species in the habitat, these will form a distinct zone, i.e., species zonation across a habitat. Species zonation also can occur in subtidal habitats in response to environmental gradients of water depth, light availability, and sediment types.

Examples of species zonation

One of the best examples of species zonation in response to tidal flat environmental gradients is afforded by mangroves and saltmarshes (Figures 4, 5, 6, and 7; and Chapman, 1938; Bridgewater, 1975; Tomlinson, 1986; Bridgewater and Cresswell, 1993; Cresswell and Bridgewater, 1998; Emery et al., 2001; Pennings et al., 2005). From mean sea level (MSL) to the high tidal mark, mangroves form zones in response to frequency of inundation, salinity of pore water, pore-water content, and sediment grain size. For instance, in a region comprised of six mangrove species (*Avicennia marina*, *Aegialitis annulata*, *Aegiceras corniculatum*, *Bruguiera exaristata*, *Ceriops tagal*, and *Rhizophora stylosa*), with strong environmental gradients of salinity and inundation, there is zonation of the species in terms of composition, vegetation structure, and plant physiognomy. The species are zoned from MSL to the high tidal mark with *Avicennia marina* found where the pore-water salinity is 40 ppt, followed by a zone of *Rhizophora stylosa* where the pore-water salinity is 45 ppt, then by a zone of *Rhizophora stylosa* and *Bruguiera exaristata* where the pore-water salinity is 55 ppt, a zone of *Ceriops tagal* where the pore-water salinity is up to 85 ppt, and finally a landward zone of *Avicennia marina* where the pore-water salinity is up to 90 ppt (Semeniuk, 1983). Likewise, various species of saltmarsh form ecological zones in response to inundation frequency and pore-water salinity on the tidal flats of estuaries.

There are similar patterns of zonation on tidal flats and in shallow-water habitats in estuaries for invertebrates such as fiddler crabs, other Brachyura, and mollusks, with the biota exhibiting zonation in response to environmental gradients (Crane, 1975; Chakraborty and Choudhury, 1985; Dittmann, 2000; Dürr and Semeniuk, 2000;



Species Zonation, Figure 9 Zonation of mollusks in an estuary across the various habitats in upper estuarine, middle estuarine, and deltaic fields (information from Semeniuk and Wurm, 2000). Species illustrating zonation are: *Hydrococcus brazieri*, *Arthritica semen*, *Acteocina* sp., *Bedeva paivae*, *Nassarius burchardi*, *Sanguinolaria biradiata*, *Spisula trigonella*, *Tellina deltoidalis*, and *Xenostrobus securis*.

Semeniuk, 2000; Bezerra et al., 2006). Species with broad environmental and habitat tolerances are able to exist within a wider range of habitats or tolerate greater fluctuation in environmental changes and thus cross over zones, while environmentally restricted species do not. For instance, within a suite of adjoining habitats varying from sand to muddy sand to mud, and varying from saline to hypersaline, or varying from subtidal to tidal flat, some species occur across habitat boundaries, while others are environmentally restricted (Figure 8). Species zonation within habitats in estuaries determined by environmental gradients is shown in Figure 9 for mollusks.

Summary

Species zonation is a common feature within estuaries where the combined and interactive effects of steep environmental gradients and food availability and to interspecies competition, herbivory, and predation result in compositional changes in biotic assemblages across these gradients. For vegetation and for rock-inhabiting biota, it is often manifest as visually distinct communities occurring in bands. For infauna, it is expressed as compositional and abundance changes in the communities across the habitat. In areas of less steep environmental gradients, species often occur in less differentiated formations, such as mottled mosaics or with diffuse zonation.

Species zonation and changes in biotic assemblages can be expressed across the whole of the estuary responding to an along-estuarine environmental gradient of salinity, from freshwater to marine, and a gradient in substrate grain size. For zonation within a given habitat, while there may be a range of factors that influence species zonation that include wave energy, tidal currents, extent of water turbidity, water depth, temperature, degree of light penetration, pH, and nutrients, some of the main environmental determinants for forcing species zonation are pore-water salinity, open-water salinity, substrate type, and inundation. These affect the survivorship of a given species, determine the suitability of a habitat for an organism, and also affect the microbiota in the environment that influence the occurrence of macrofauna and macroflora. One of the best examples of species zonation in estuaries is afforded by mangroves in tropical regions and saltmarsh on tidal flats in temperate regions in response to tidal flat environmental gradients. For example, in mangroves with strong environmental gradients of salinity and inundation, there is zonation of the species in terms of composition, vegetation structure, and plant physiognomy.

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Cross-references

Mangroves
Saltmarshes
Tidal Flat Salinity Gradient

SPIT

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Synonyms

Barrier spits; Headland spits; Sandspit

Definition

A spit is a coastal landform, a depositional ridge, or an embankment of sediment (Evans, 1942) with one end attached to a headland of the coast that serves as the source of sediment (proximal end) and the other end extending into open water (distal end). It is younger than the headland to which it is attached.

Description

Although spits can form and are maintained in a variety of environmental settings, they develop most readily in large lakes and wave-dominated coasts with a small tidal range, which provides optimum conditions for undisturbed spit development. Offshore waves normally approach the surf zone of a coast at an oblique angle. A combination of shore-oblique swash caused by the incoming waves and shore-normal backwash caused by gravity creates a longshore drift of sediment which is further strengthened by longshore currents generated by wave breaking. Sediment is entrained by strong turbulence induced by wave breaking, and it is transported down-drift along the coastline by longshore currents. Longshore sediment transport rate remains constant if a uniformity of waves and nearshore isobaths exist along the coastline (USACE, 1984).

Net deposition of sediment occurs where the longshore uniformity is broken by a decrease of wave energy. This is normally caused by a deepening of the bathymetry or a change of the coastline orientation. In the latter case, the boundary constraint of the longshore currents by the coastline no longer exists, and the currents are veered by a barotropic pressure induced by wave radiation stress. On the side to which the currents are veered, turbulence is dissipated by calm waters which cannot entrain the full load. Much of the sediment is deposited as a result, forming a submerged bar. This submerged bar subsequently acts to maintain the original direction of the longshore currents and also serves as a reservoir for sediment accumulation. Deposition of sediment on the submerged bar will not cease until a uniformity of waves and nearshore isobaths is again achieved. Eventually a spit develops above water by this process.

When the submerged bar in front of a spit expands, it may cause significant wave refraction which deflects the longshore currents around the distal end of the spit to form a hook or a secondary recurved spit. A change in prevailing wind direction may also cause similar effects. Wave refraction in multiple directions often induces the formation of a complex spit system.

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Cross-references

Bar
Barrier Spits
Beach Processes
Coastal Barriers
Coastal Cliffs
Coastal Landforms
Headland Breakwaters
Sediment Resuspension
Wave-Driven Sediment Resuspension

STORM SURGES

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Synonyms

Inundation; Surge

Definition

Storm surge refers to the increased water level above the predicted astronomical tide due to storm winds and atmospheric pressure changes.

Storm surge is principally produced by a storm's wind stress that pushes water in the same direction as the wind (CHL, 2013). Low atmospheric pressure associated with the storm makes a minor contribution to the overall storm surge. Storm surge is calculated by subtracting the astronomical tide from the observed water level during the storm. It is considered the most significant hazard to life and property, as the increased water level permits large waves to break inland of the "normal" surf zone. Storm surge is complex and dependent upon many contributing factors (NHC, 2013) including:

Intensity: The greater the wind speed, the greater the wind stress and surge. Strengthening or weakening of the storm before landfall will play a significant role in the overall surge.

Forward Speed: In a cyclonic storm, the relative wind speed at any point is the vector sum of the local wind speed and the forward speed of the storm. For example, a cyclonic storm with a maximum wind speed of

80 mph and forward speed of 50 mph will have relative winds of 130 and 30 mph on opposite sides of the storm.

Timing: The time of the storm with relation to the tide makes a significant impact on the magnitude of storm surge. For example, consider an area with an 8 ft normal tide range. If storm surge is 10 ft, at low tide the additional water level would only be 2 ft above the normal high tide elevation; however, if the surge occurs at high tide, the water level would be 10 ft above the normal high tide elevation at that time.

Storm Size: The greater the storm size (radius of maximum winds speed), the larger the surge and area the surge will impact.

Angle of Approach: The angle of the storm path relative to the coast will play a significant role in determining the overall surge. Comparing two similar storms, the surge from a storm moving along the coast will be significantly less than one making direct landfall.

Central Pressure: For significant storms (i.e., hurricanes), the central pressure on the water surface makes a small but finite contribution to the overall surge.

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Cross-references

Extratropical Storms
Extreme Events (Hurricanes)

STRATIGRAPHY OF ESTUARIES

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Definition

The stratigraphy of estuaries is the vertical and lateral array of sediments that occur in smaller-scale environments within an estuary, as well as longitudinally and transversely across the whole of an estuary.

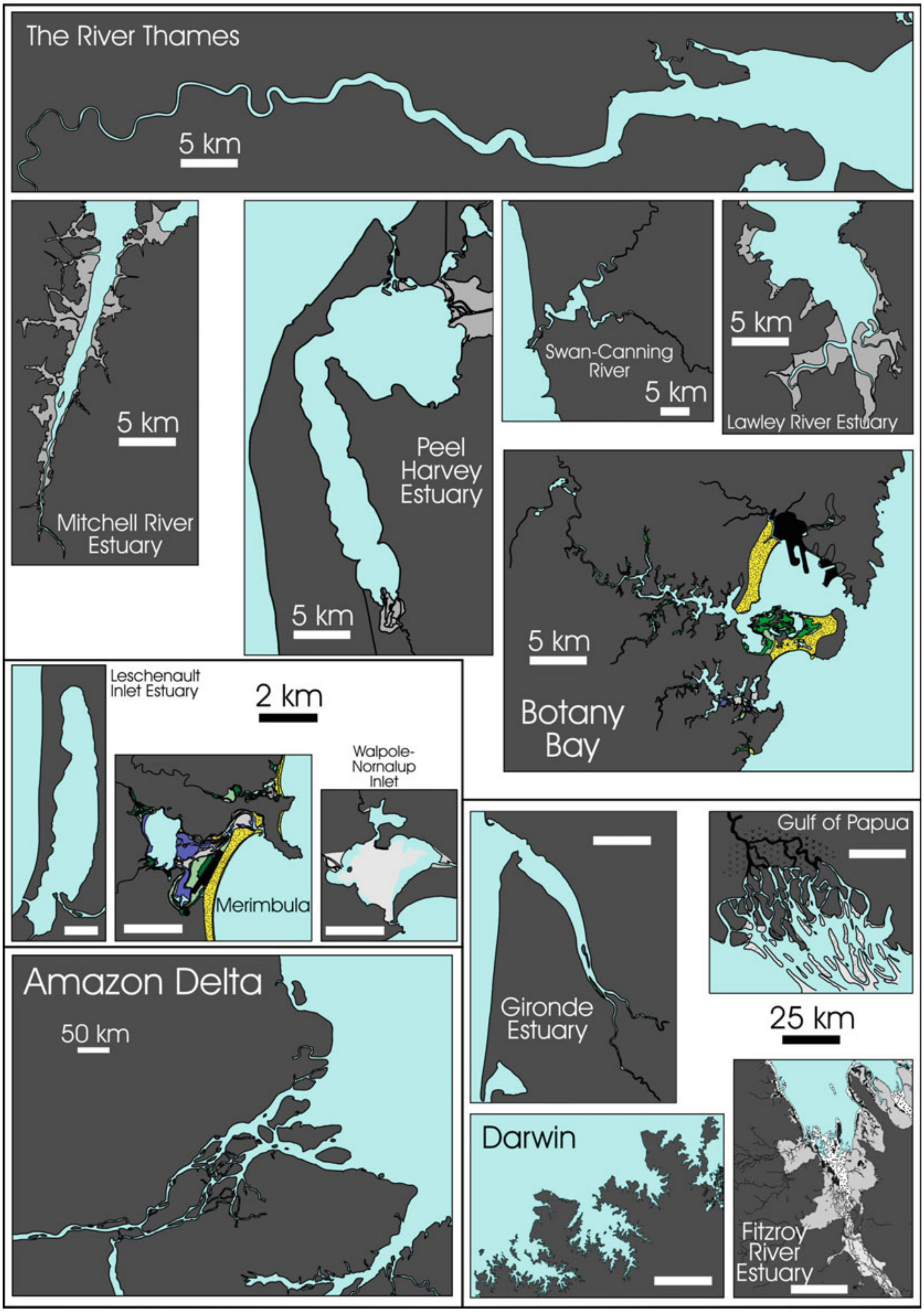
Stratigraphy of estuaries: overview and settings

Estuarine stratigraphy can be viewed at two scales: firstly, the environmentally distinct small-scale stratigraphy, reflecting processes and products of internal estuarine geomorphic units and their sedimentary environments, and, secondly, at the larger scale, the stratigraphy for the whole estuary that represents the amalgamation of small-scale stratigraphic suites into estuarine-longitudinal and estuarine-transverse mosaics. At both scales, there are a range of environmental factors and processes which contribute to a striking spatial variability of stratigraphy. Prior to describing the small-scale stratigraphy and the lithologic detail of the large-scale stratigraphy, the setting of estuaries and the processes and determinative factors leading to sediment generation and accumulation are described to provide an understanding of the development of sedimentary and stratigraphic suites.

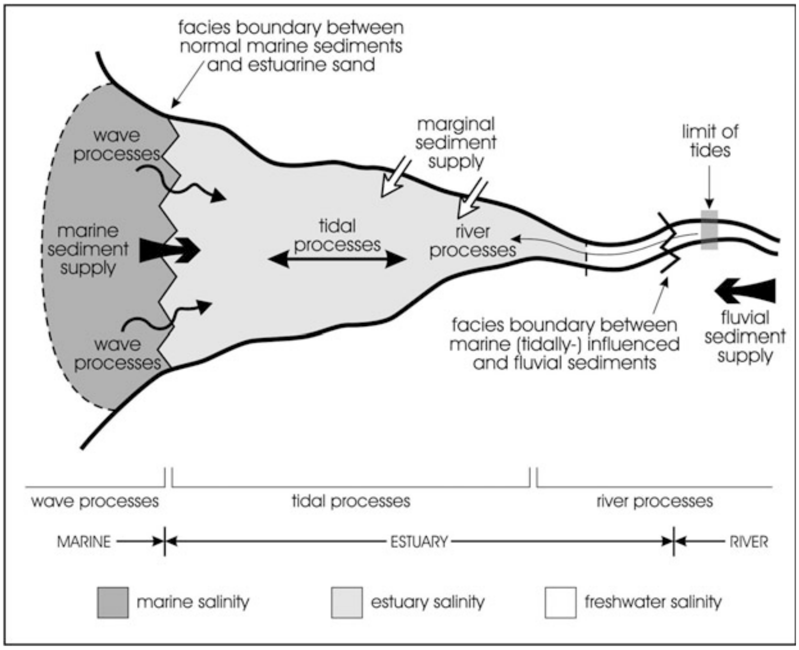
The landform settings and origin of estuaries are variable, ranging from incised valleys, such as rias and fjords, to flooded valleys on coastal plains, to barred rias, and to barred coastal plain lagoons that have a riverine input and an ocean outlet. Depending on their original geomorphology, the extent of inundation by the Holocene postglacial transgression, and local wave and wind energy, the forms of estuaries range from open estuarine bays, to narrow estuarine bays or gulfs, to inlets that are partly barred or nearly fully barred basins. Estuaries also occur in a wide spectrum of climates and reside in different coastal energy settings dominated either by waves, tides, or wind or combinations of the three. Sediments derive from a range of drainage basin types and sizes, and from different source rock provenances in different climatic settings, resulting in variable volumes of sediment influx, and different types and sizes of sediment particles. As such, estuaries exhibit diverse attributes such as size and shape, sediment sources, sediment influx rates, hydrodynamic/sedimentary dynamics, and, of course, stratigraphic packages. Reflecting this variability in setting, coastal and estuarine processes, and estuarine form (Figure 1), there are a wide range of stratigraphic packages that can occur in estuaries.

One of the settings in which estuaries also occur is in the outlet mouths of distributary channels of large marine deltas, but in this context, the stratigraphy therein is that of the enclosing delta (see Gould, 1970; Coleman et al., 1970; Allen, 1970; Reineck and Singh, 1980) and will not be described here.

The essence of an estuary is that it is a river-to-marine transitional environment where marine salinity is measurably diluted by (riverine) freshwater in a valley tract, an inlet, a coastal lagoon, or an embayment, producing a salinity gradient from the river to the sea. In terms of hydrochemistry, biota, and processes, there are a riverine component toward the landward part of an estuary and a marine component toward the seaward part (Figure 2). This salinity gradient from river to sea is mirrored in the gradients in sedimentary processes, sediment types, facies, and stratigraphy. While this is a hydrochemical perception, geologists have emphasized the sedimentologic/stratigraphic aspects of an estuary (Dalrymple et al., 1992), viewing the basin of an estuary as a sedimentary sink for sediments deriving from both



Stratigraphy of Estuaries, Figure 1 Diagram showing a range of estuaries in terms of geometry and scale. These estuaries also occur in a range of different climates (from tropical to temperate) and oceanographic settings (tide-dominated vs. wave-dominated). Some of the estuaries shown here are used later to illustrate the variety and styles of stratigraphic fills, viz., the Thames, the Gironde, the Peel-Harvey Estuary, the Lawley River Estuary, the Swan-Canning Estuary, the Walpole-Nornalup Estuary, the Leschenault Inlet Estuary, and the Fitzroy River Estuary.



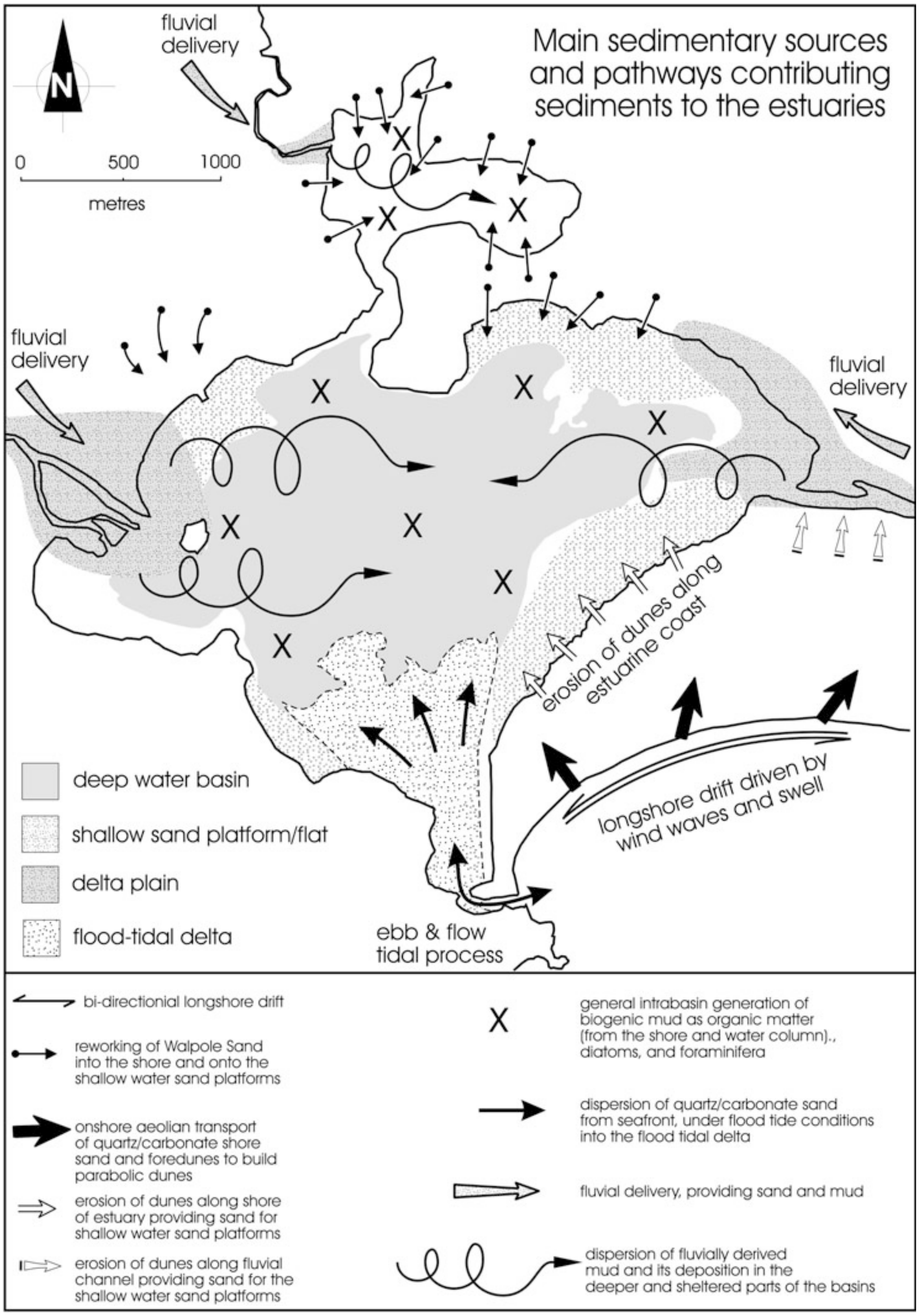
Stratigraphy of Estuaries, Figure 2 A typical funnel-shaped estuary with its hydrochemical fields and marine, estuarine, and fluvial processes that occur longitudinally along its length that determine the sedimentologic processes and responses (modified after Dalrymple et al., 1992).

fluvial and marine sources and containing facies influenced by tide, wave, and fluvial processes. In this context, the essence of an estuary is that it is a river-to-marine transitional environment, which is reflected in the change from riverine conditions (with its assemblage of lithofacies) to marine conditions (with its assemblage of marine lithofacies), with the estuarine basin being a unique assemblage of facies (Figure 2).

Sedimentologically, an estuary acts as a basin, semi-protected or nearly fully protected from the sea. Therein, fluvial sediment can be delivered and largely trapped. Marine processes deliver sediment at its seaward portions in an open bay setting, or by marine coastal transport processes though narrow inlets, or by washover across a low barrier. Estuarine processes within the estuarine basin itself operate to develop intra-basinal sediment, transport and disperse sediment, and develop sedimentary suites from the materials delivered from fluvial, marine, and intra-basinal estuarine sources. The magnitude of the tidal range for the region where the estuary resides, and/or the extent that the shape of the estuary magnifies the tidal range, will determine how far upstream tidal effects are experienced and to what degree tidal patterns will influence sedimentation patterns (Figure 2).

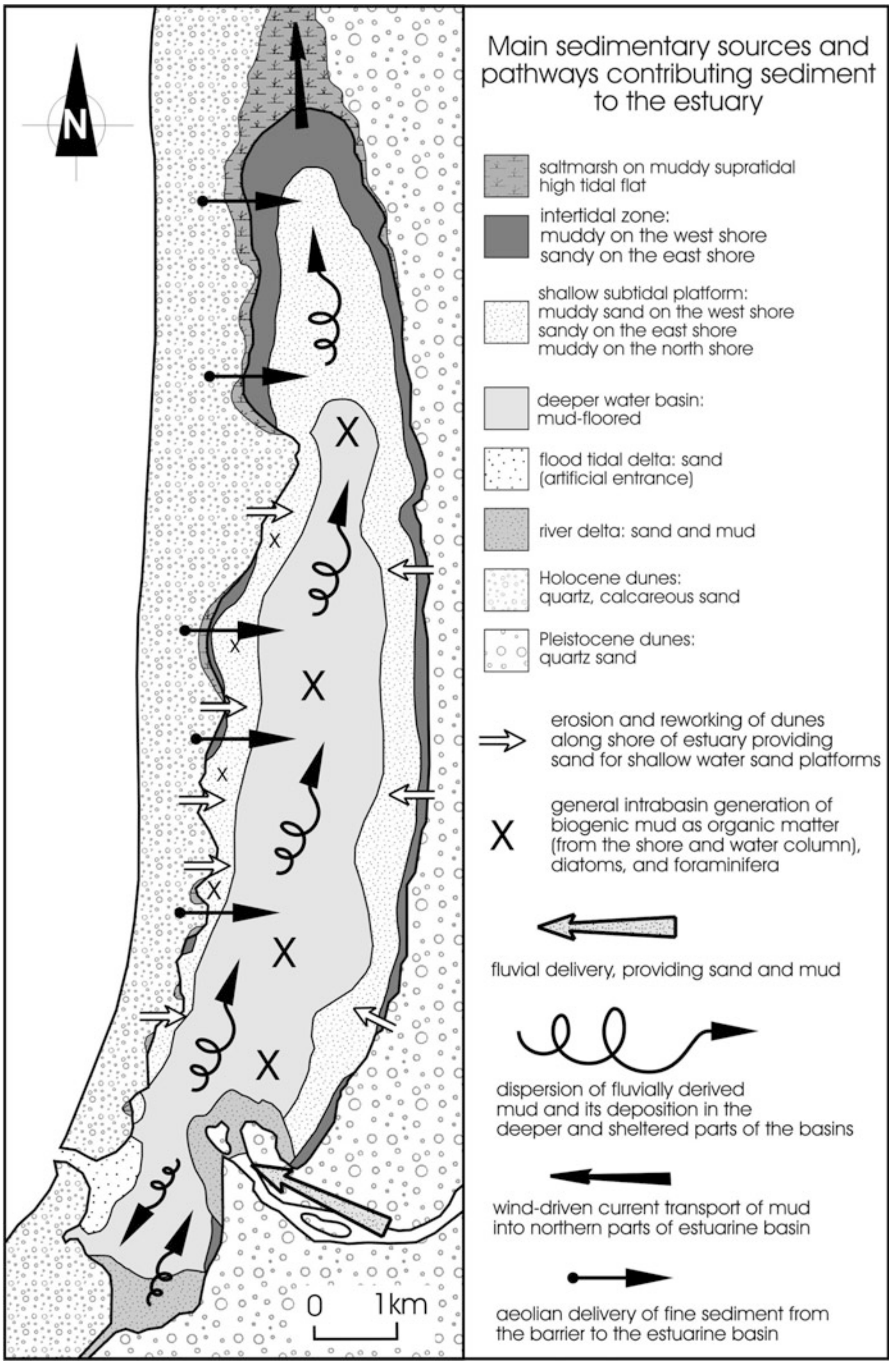
The sources and types of sediment that build stratigraphic sequences, their delivery system to the estuary, and where the sediment finally is emplaced in the estuary are described below in four systems, viz., the riverine system, the central estuary, the marginal estuarine system, and the marine system.

The riverine system is usually a shallow water system. By channel flow and floods, it delivers sand (usually quartz sand), terrigenous mud (usually clay minerals and quartz silt), and gravel to the estuary. While these sediments are mainly located in the estuary headwaters, they can be dispersed into the central parts of the estuarine basin, graded in grain size from coarsest sand at the deltas and river mouths to fine and very fine sand away from the deltas. Riverine mud is the sediment type that is most widely dispersed, and, since it is carried in suspension, it can be deposited some distance from the river mouth. The central estuary is a shallow water to moderately deepwater system. The central estuary generates sand (as foraminifera, algal fragments, and fragmented invertebrate skeletons), gravel (as invertebrate skeletons), mud (as comminuted thin-shell fragments, disintegrated algal skeletons, diatoms, and sponge fragments), and biogenically built structures such as biostromes (e.g., mussel beds), bioherms (e.g., oyster reefs and worm-tube reefs), and weed-built, weed-constructed, or weed-trapped sediment sheets. These particle types and biogenic sedimentary products are mainly located in the central part of the estuary, while plant products and biostromes/bioherms specifically develop in the shallow water marginal parts of the estuary. However, through intra-estuarine transport processes, the mud-sized and sand-sized particles listed above can be dispersed into deeper water parts of the central part of the estuary. As with fluvial mud, mud-sized particles generated within the estuary are the most widely dispersed.



Stratigraphy of Estuaries, Figure 3 Main sedimentary sources and pathways of sediment movement within the Walpole-Nornalup Inlet Estuary, a twin ria system with three river inputs, illustrating the principles of sediment particle sources and transport.

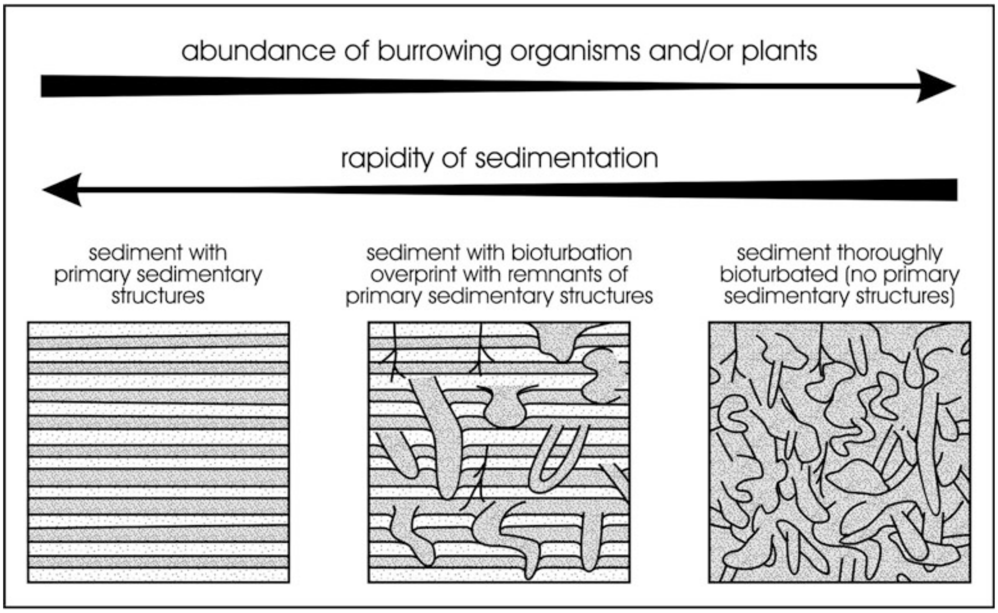
The central estuary is also the location where mud delivered by rivers accumulates, because it is generally the deepest part of the estuary and a low-energy sink. In addition, the central basin is the site for accumulation of organic matter generated by shallow water biotic assemblages that might have been suspended in the water column and transported into this low-energy area. As such, the sediments accumulating in the central estuary range from muds to organic-matter-enriched muds. Without bioturbation, the muds in the central estuary are laminated. However, more generally, the muds in the central estuary are thoroughly bioturbated. The central estuary is also



Stratigraphy of Estuaries, Figure 4 Main sedimentary sources and pathways of sediment movement within the Leschenault Inlet Estuary, an elongate shore-parallel estuarine lagoon barred by a dune barrier, and with river inputs restricted to its southern end, to illustrate the principles of sediment particle sources and transport.

the site for accumulation of aeolian very fine sand and silt, usually deriving from dune barriers seaward of the estuary. In these circumstances, if the muds of the central estuary are laminated, particle-width laminae of aeolian sand and silt define some of the lamination.

The marginal estuarine system is a shallow water to geomorphically emerged system. It receives exogenic sediment from a number of sources. Sediment may be reworded and delivered from adjoining uplands, other supratidal locations, or alongshore from elsewhere in the



Stratigraphy of Estuaries, Figure 5 The progressive obliteration of primary sedimentary structures in sediments reflecting the relative balance between biota abundance and the rapidity of sedimentation. The primary sedimentary structures, once diagnostic of an environment, can be reduced firstly to root-structured or burrow-structured sediments (within which vestiges of the former structures are discernable) and then finally to a thoroughly bioturbated sediment (in which only grain sizes, grain types, and the presence of biota can be determined for use as environmental indicators).

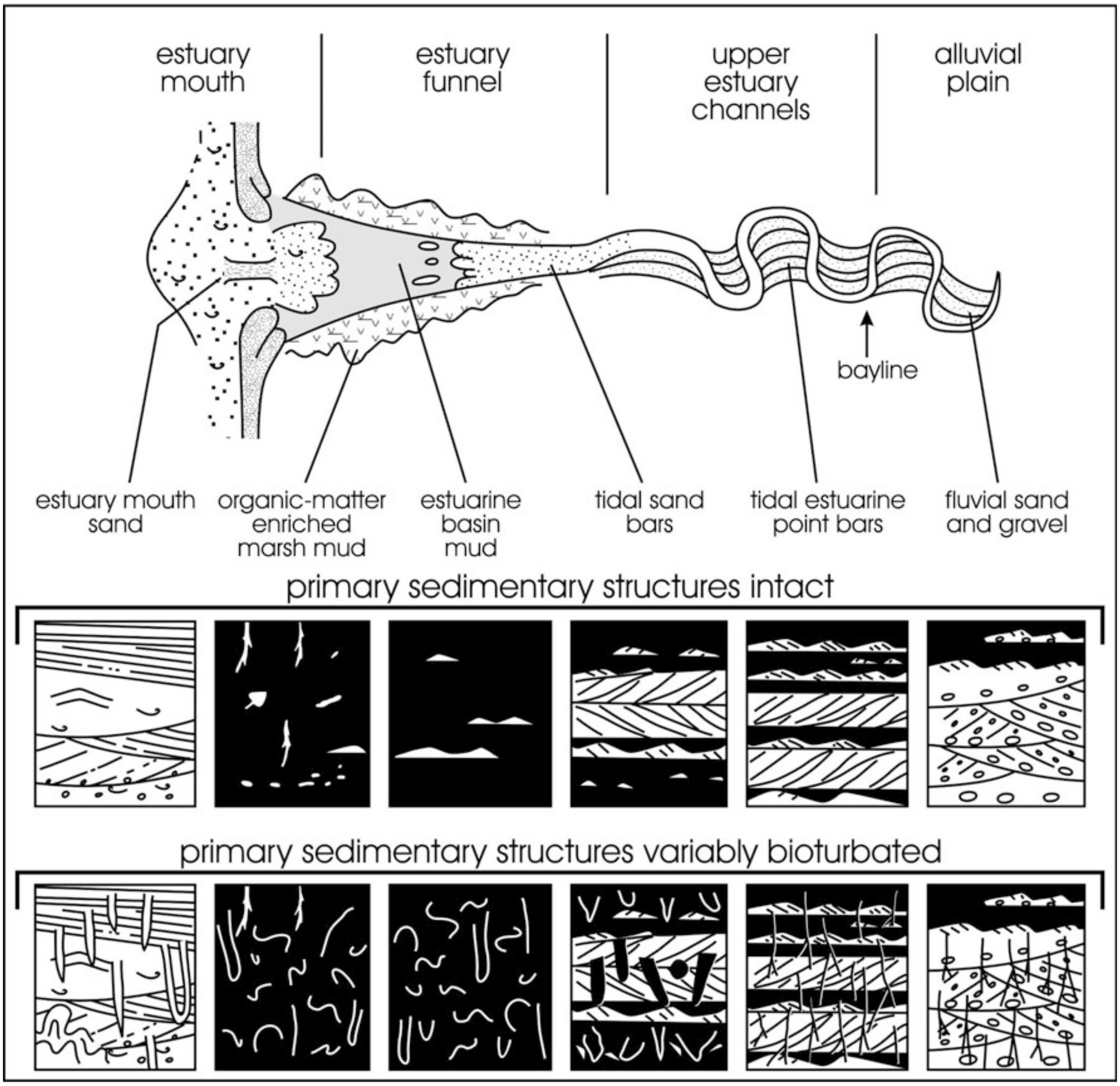
estuary (as sand, mud, or gravel), or it may consist of peat from plants inhabiting shorelines and marginal shallow water environments, or carbonate mud (in marginal lagoons) generated from algal meadows, or mud brought in by suspension on the high tide or by storms. Depending on whether the surrounding uplands are rocky, preexisting older sedimentary deposits, or stranded estuarine deposits, the material reworked by sheetwash, shoreline erosion, or fluvial action may be lithoclast (rock) gravel, sand (usually quartz), and mud. These sediments are shed onto the supratidal to high-tidal parts of the estuary. The tidal to shallow subtidal marginal estuarine environments, through prevailing wave action, storm, and tide activity, generally comprise a sediment platform underlain by sand and (in low-energy settings) mud. The shallow water biota thereon contributes shell and fragments, algal fragments, diatoms, and plant organic matter.

The marine system is usually a shallow water environment. The marine system delivers sand (as quartz and marine skeletal material, viz., foraminifera, algal fragments, and fragmented invertebrate skeletons) and shell gravel to the estuary. These particles are transported as sand sheets by waves, tides, and storms, across and into the entrance of the estuary, or are funneled by waves and tides through the narrow tidal inlet, or multiple inlets (that breach the barrier), to form radiating to palmate plume(s) of sediment (the flood-tidal delta). Marine sediment is also transported during storms into the estuary across low barriers to form washover lobes or is reworked from the leeward barrier by estuarine waves and tides to form shoreline spits, ribbons, or platforms on the leeward margin of the barrier. The tidal-delta plumes and washover lobes encroach into the estuary, often migrating into and overlying the sediments of the deeper water basin.

The variety of sediment sources and processes that emplace the particle types and sediment types in an estuary is shown in a case study of the Walpole-Nornalup Inlet Estuary and the Leschenault Inlet Estuary both in Western Australia (Figures 3 and 4). In the former, the surrounding uplands are weathered Precambrian rock, Cainozoic quartz sand, and a Quaternary dune barrier, and the rivers deliver sand and mud, and there is an intra-basinal production of biogenic particles. In the latter, the uplands are Pleistocene dune sands and a Holocene dune barrier, and the rivers deliver sand and mud, and there is an intra-basinal production of biogenic particles. These examples provide case studies of sediment sources and pathways, firstly, in a barred ria system where there are three sites of river input, though the riverine input is axial, and, secondly, in a barrier-and-lagoon estuary system where river input is at one end of the estuarine lagoon.

Small-scale local environment stratigraphy

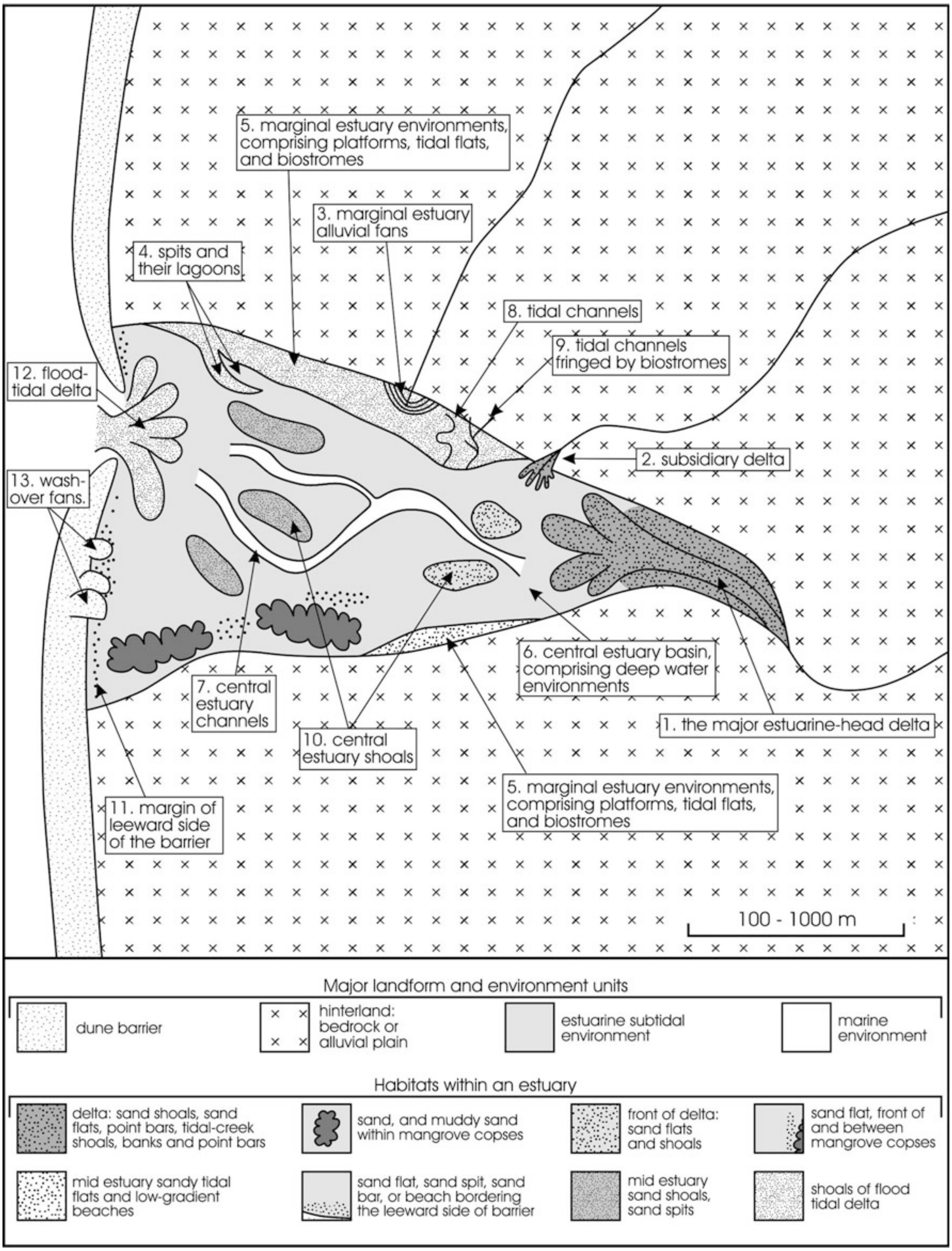
In the estuarine environment where the sediment either is delivered exogenically or is generated intra-basinally, the local processes of wave action, tidal currents, fluvial currents, wind, hydrochemical processes, shoreline freshwater seepage, and biogenic activity (exoskeleton and



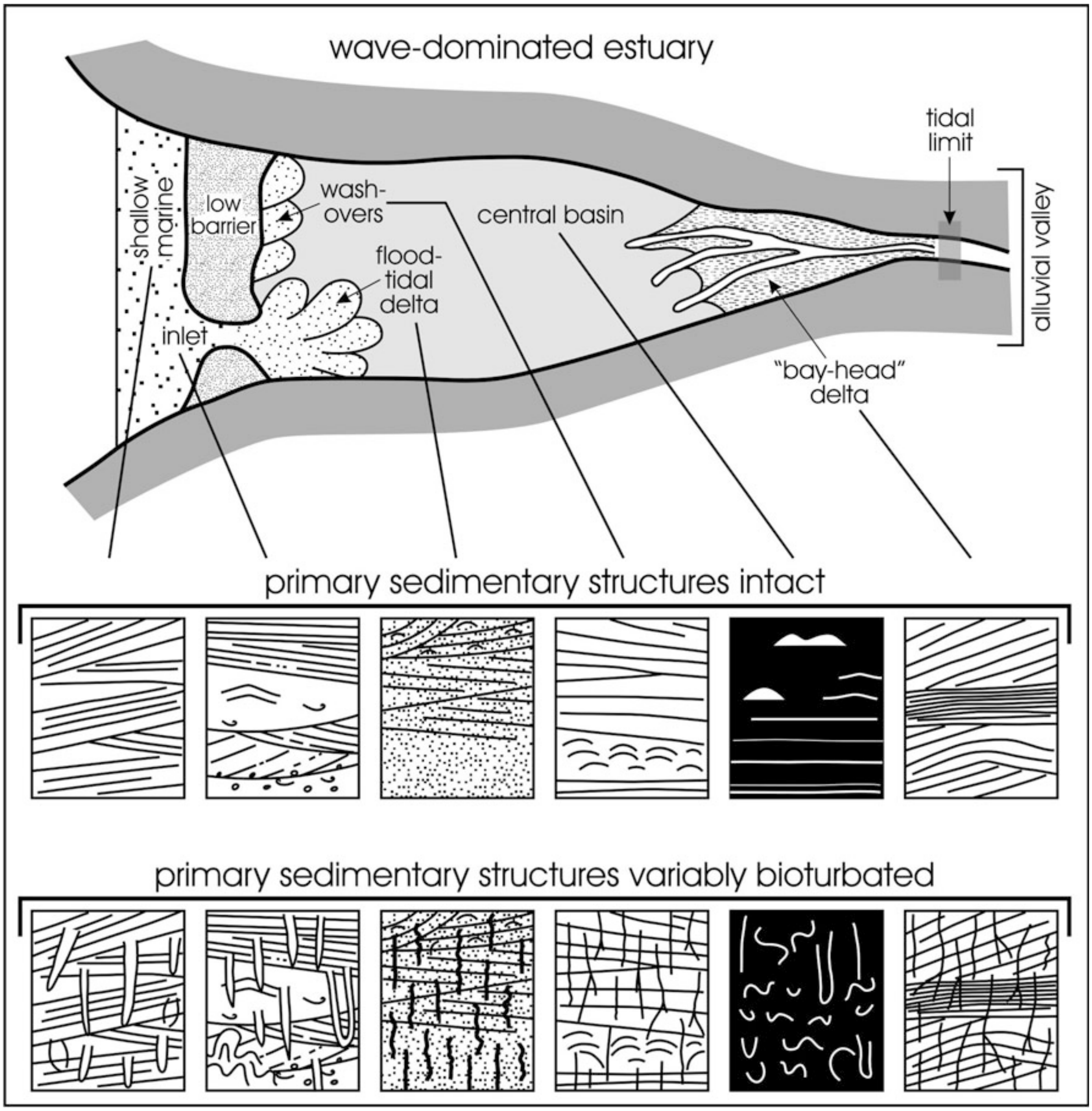
Stratigraphy of Estuaries, Figure 6 The lithologies and sedimentary structures in the various sub-environments of a tide-dominated estuary exemplified by an estuary such as the Gironde (Adapted from Allen and Posamentier, 1993). This diagram extends the results of Allen and Posamentier (1993) for a Gironde-type estuary in that, where there is benthos abundant in the aquatic and tidal sub-environments/facies and vegetation inhabits tidal and riverine sub-environments/facies, the effects of bioturbation have been added.

endoskeleton production generating shell gravel, skeletal sand, and biostromes and reefs, plant production, bioturbation, and biomediated mineralization) result in a variety of sedimentary processes and products. These include biogenic particles such as mud, sand, and gravel; partitioning of grain sizes of particles of both exogenic and intra-basinal origins; dispersal/transport of sediment; shaping of bedforms and generation of sedimentary structures by wave action and tidal current; bioturbation and root structuring; accumulation of sediment with its signature lithology for a given facies; and (with lateral progradation and vertical accretion) development of facies-specific stratigraphic packages.

A significant factor in the development of facies-specific lithologies and their potential as diagnostic features in identifying environments is the extent to which bioturbation may overprint primary sedimentary structures. Many primary sedimentary structures such as lamination, wavy lamination, flaser bedding, and climbing ripple structures, among others, are diagnostic indicators of hydrodynamic conditions and formative environments (Reineck and Singh, 1980) and can be used to identify specific estuarine environments. However, bioturbation by animals and/or plants can destroy the sedimentary structure evidence, reducing various individual diagnostic lithologies to a similar appearance. Figure 5 shows the



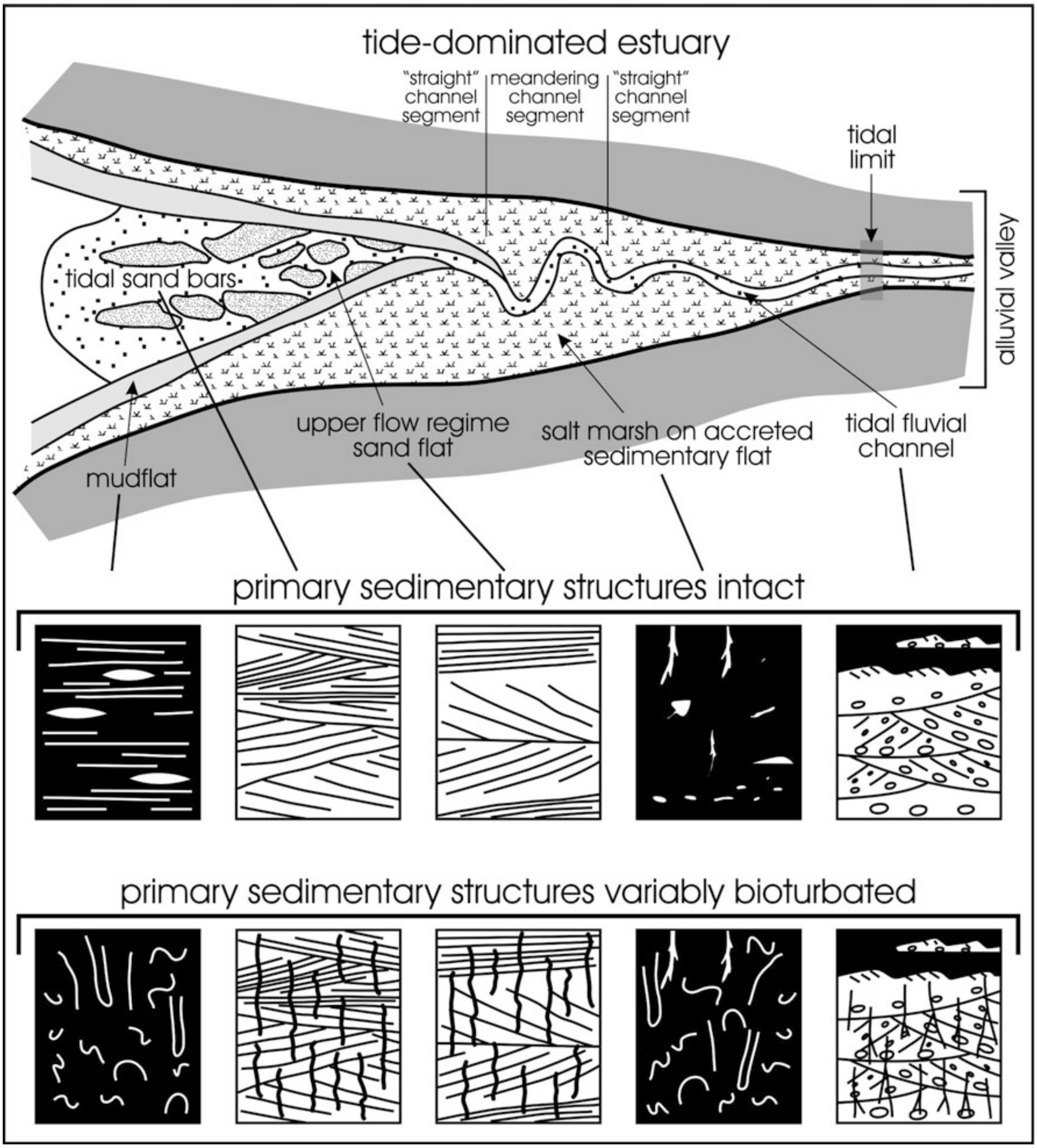
Stratigraphy of Estuaries, Figure 7 The range of sub-environments/facies that may occur in an estuary. The diagram is not specific to a fluvial-, wave-, or tide-dominated estuary, but is generalized. Each of these sub-environments/facies may/will have diagnostic sedimentary features and biota.



Stratigraphy of Estuaries, Figure 8 The idealized diagnostic small-scale sedimentary features and their bioturbated equivalents of the different sub-environments/facies of a wave-dominated estuary (Diagram based on estuarine model of Dalrymple et al. (1992), with approach of Allen and Posamentier (1993), for illustrating the character of the small-scale facies).

process whereby distinct sedimentary structures that may be or are diagnostic of specific environment are progressively obliterated by animal bioturbation or by root structuring. Bioturbation overprinting and obliterating primary sedimentary structures is typical in shoreline estuarine environments in humid climates where there is much vegetation or in tropical climates where, in addition to shoreline vegetation, there is a diverse and abundant benthos. In this context, the illustrations that follow are shown as small-scale primary sedimentary structures and also with the overprint of bioturbation because, depending on the occurrence and abundance of biota, either end product can eventuate.

As noted earlier, viewed in an overall context, an estuary can be wholly river-dominated, wave-dominated, or tide-dominated or can be river-dominated at its upstream part and wave-dominated or tide-dominated at its seaward part. The extent that an estuary is dominated by river, wave, or tide processes and where these environments occur will determine the nature of small-scale stratigraphy

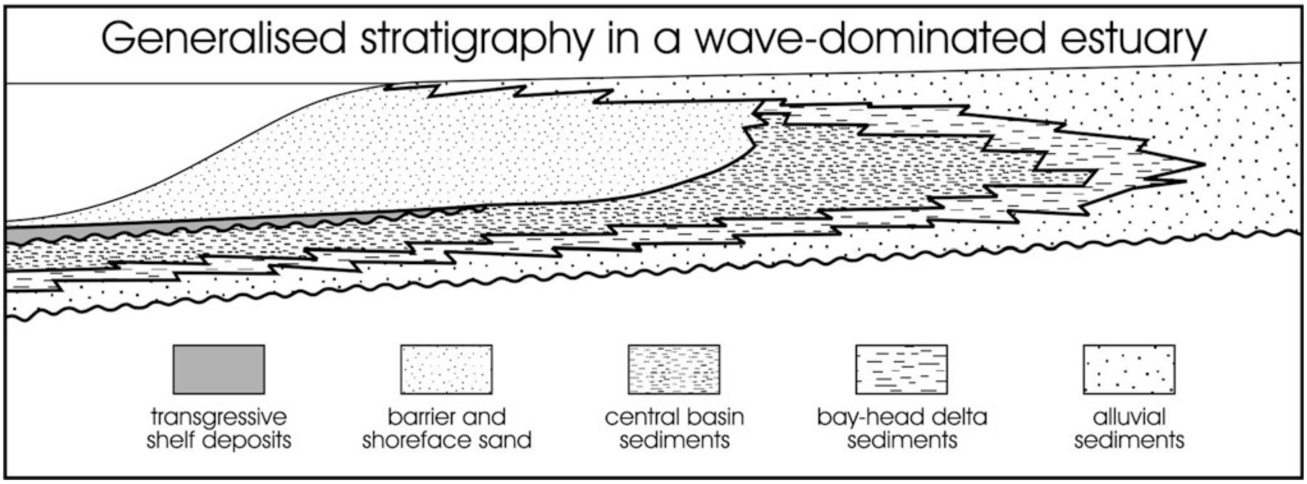


Stratigraphy of Estuaries, Figure 9 The idealized diagnostic small-scale sedimentary features and their bioturbated equivalents of the different sub-environments/facies of a tide-dominated estuary (Diagram based on estuarine model of Dalrymple et al. (1992), with approach of Allen and Posamentier (1993), for illustrating the character of the small-scale facies).

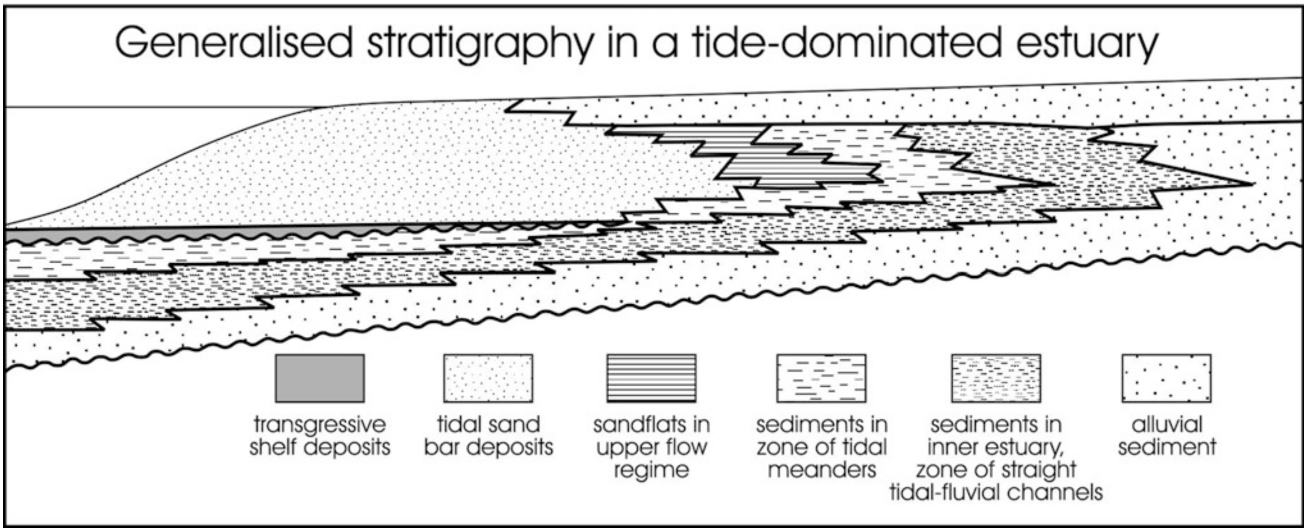
at a given site. In this context, for instance, using the Gironde Estuary from France as a model, Allen and Posamentier (1993) highlighted the facies types in terms of lithology and sedimentary structures along tide-dominated estuaries and the small-scale stratigraphy diagnostic of each sub-environment within such estuaries (Figure 6).

The main sedimentary environments, or facies, that occur within an estuary from landward to seaward are as follows (Figure 7):

1. The major estuarine-head delta(s)
2. Subsidiary deltas
3. Marginal estuary alluvial fans
4. Spits and their lagoons
5. Marginal estuary environments, comprising platforms, tidal flats, mangrove- and/or salt marsh-vegetated tidal flats, and biostromes
6. Central estuary basin, comprising deepwater environments
7. Central estuary channels



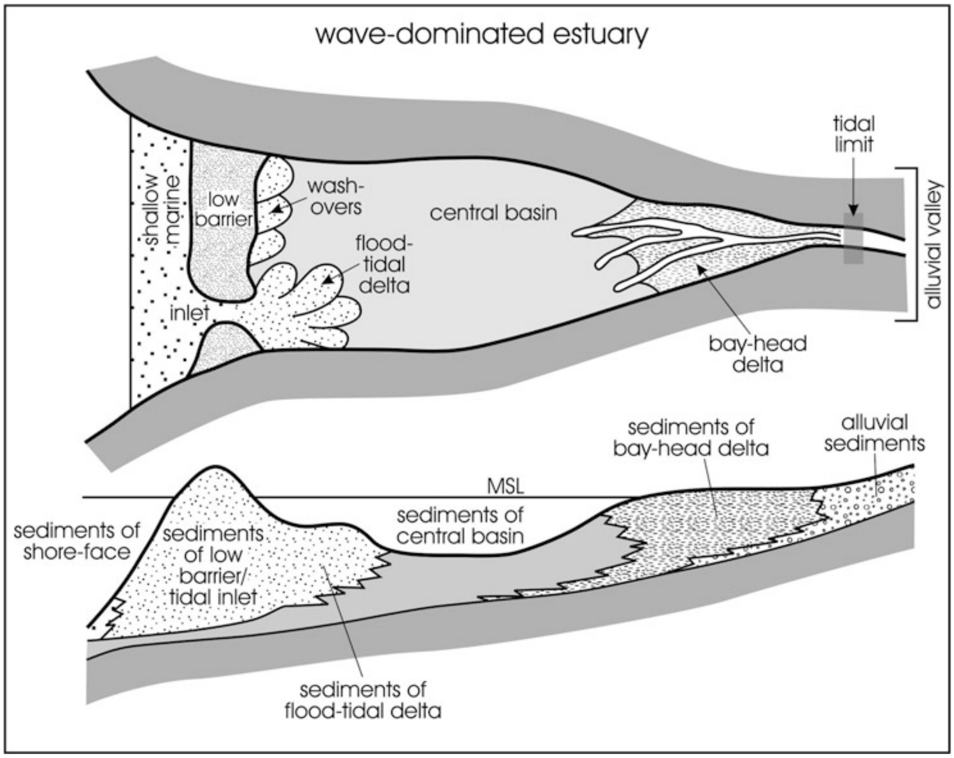
Stratigraphy of Estuaries, Figure 10 The idealized whole-of-estuary down-valley-tract longitudinal stratigraphy in a wave-dominated estuary showing transgressive and progradational relationships (Diagram simplified and modified from Dalrymple et al., 1992). While the lithologic suites related to setting in the estuary are generalized and not lithology specific, the lithologic details of the various large-scale stratigraphic units will be determined by climate setting, wave energy, sediment types supplied, and biota.



Stratigraphy of Estuaries, Figure 11 The idealized whole-of-estuary down-valley-tract longitudinal stratigraphy in a tide-dominated estuary showing transgressive and progradational relationships (Diagram simplified and modified from Dalrymple et al., 1992). While the lithologic suites related to setting in the estuary are generalized and not lithology specific, the lithologic details of the various large-scale stratigraphic units will be determined by climate setting, tidal range, sediment types supplied, and biota.

8. Tidal channels
9. Tidal channels fringed by biostromes
10. Central estuary shoals, some vegetated by mangrove and/or salt marsh
11. Margin of leeward side of the barrier
12. Flood-tidal delta
13. Washover fans

Each of these environments (or estuarine sub-environments or facies) listed above has not relegated to a setting of a wave-dominated or a tide-dominated estuary but rather to a “general estuary.” However, these types of environments (or estuarine sub-environments) or facies generate specific suites of sediments and, with accretion, a specific stratigraphy. This expression of stratigraphy at the specific environmental level or the level of geomorphic unit is small-scale stratigraphy, which focuses on the details of sedimentary structures, lithologies, lithologic interlayering, and biota imprints preserved/evident within a given facies, estuarine sub-environment, or geomorphic unit.

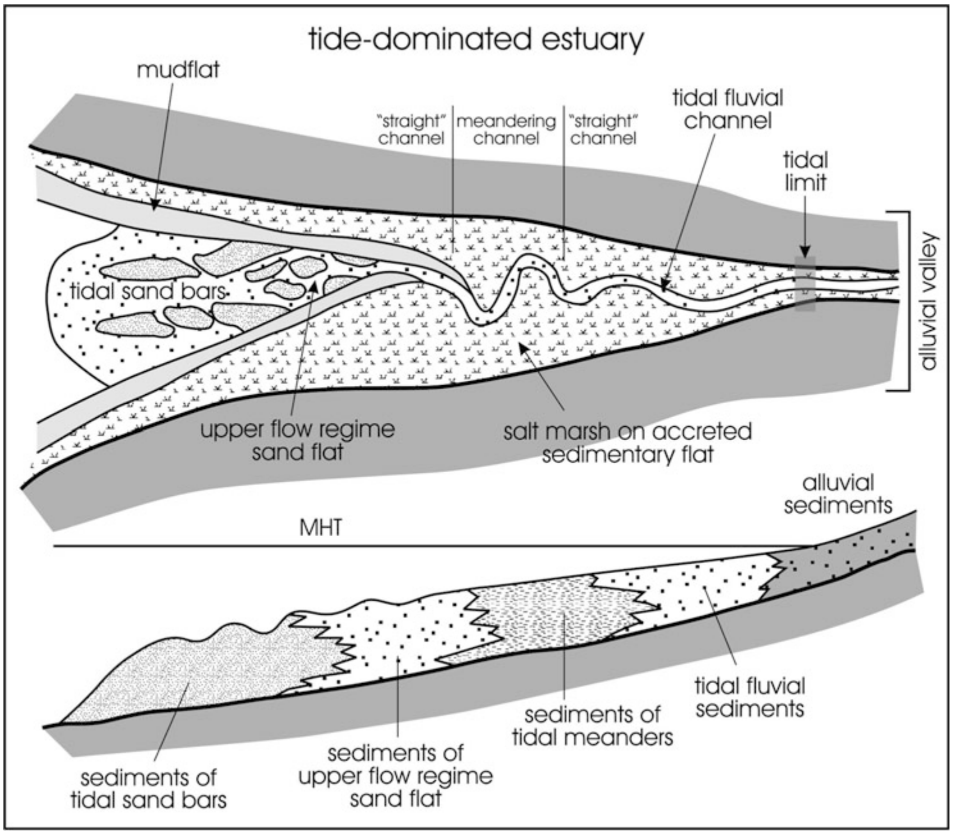


Stratigraphy of Estuaries, Figure 12 The idealized whole-of-estuary stratigraphy in a wave-dominated estuary showing lithofacies types according to location within the estuary, but not as a progradational sequence (Diagram modified from Dalrymple et al., 1992). Again, while the lithologic suites related to setting in the estuary are generalized and not lithology specific, the lithologic details of the various large-scale stratigraphic units will be determined by climate setting, tidal range, sediment types supplied, and biota.

The overall estuarine geometry, as well as more specifically the mid-estuary geomorphology and processes, determines how facies within an estuary are formed and how they are interrelated, are juxtaposed, or overlap at the large scale. For instance, an estuarine basin that is semi-closed or meandering, both with limited fetch, will not generate intra-estuarine wind waves nor be subject to ocean swell and waves that can penetrate into the estuary. Such estuaries will be dominated by fluvial processes and tidal processes, depending on whether tides are macrotidal or microtidal and how far the tides penetrate into the riverine sector. However, estuaries with a large surface water area, even though barred or semi-enclosed, if located in a region of strong winds, can be wave-dominated internally because of the large intra-estuarine fetch. Estuaries that are funnel-shaped and open to the sea in the appropriate orientation can be wave-dominated because they receive swell and wind waves from the open marine environment that penetrate far into the estuary.

Small-scale stratigraphy at any given site is influenced by the extent to which an estuary is river-dominated, wave-dominated, or tide-dominated and where these environments occur. Figures 8 and 9 illustrate the idealized small-scale sedimentary features that are diagnostic of the different sub-environments of a wave-dominated and a tide-dominated estuary, respectively, and the small-scale sedimentary features of each where bioturbation has become dominant.

Small-scale stratigraphy is also affected by the climate setting of the estuary itself in that climate influences the development of lithology, particularly in the tidal and supratidal environments, where facies have become emergent (shoaled) to levels of the highest tide or to levels above high tide (supratidal) by storm sedimentation and inhabited by vegetation. The climate setting also can determine, for instance, the biodiversity and abundance of shell material contributed to the estuarine facies, and the extent and type of marginal facies that will be developed and, thirdly, for stratigraphic sequences that have shoaled to tidal and supratidal levels, whether the facies that cap the sequences are peat (in humid climates), root-structured lithologies formed in salt marsh environments, such as root-structured and bioturbated sand, muddy sand, or mud (in humid to subhumid climates), or root-structured, bioturbated, desiccated sand, muddy sand, or clay-mineral mud (in semiarid to arid climates). Also, if



Stratigraphy of Estuaries, Figure 13 The idealized whole-of-estuary stratigraphy in a tide-dominated estuary showing lithofacies types according to location within the estuary, but not as a progradational sequence (Diagram modified from Dalrymple et al., 1992). Again, while the lithologic suites related to setting in the estuary are generalized and not lithology specific, the lithologic details of the various large-scale stratigraphic units will be determined by climate setting, tidal range, sediment types supplied, and biota.

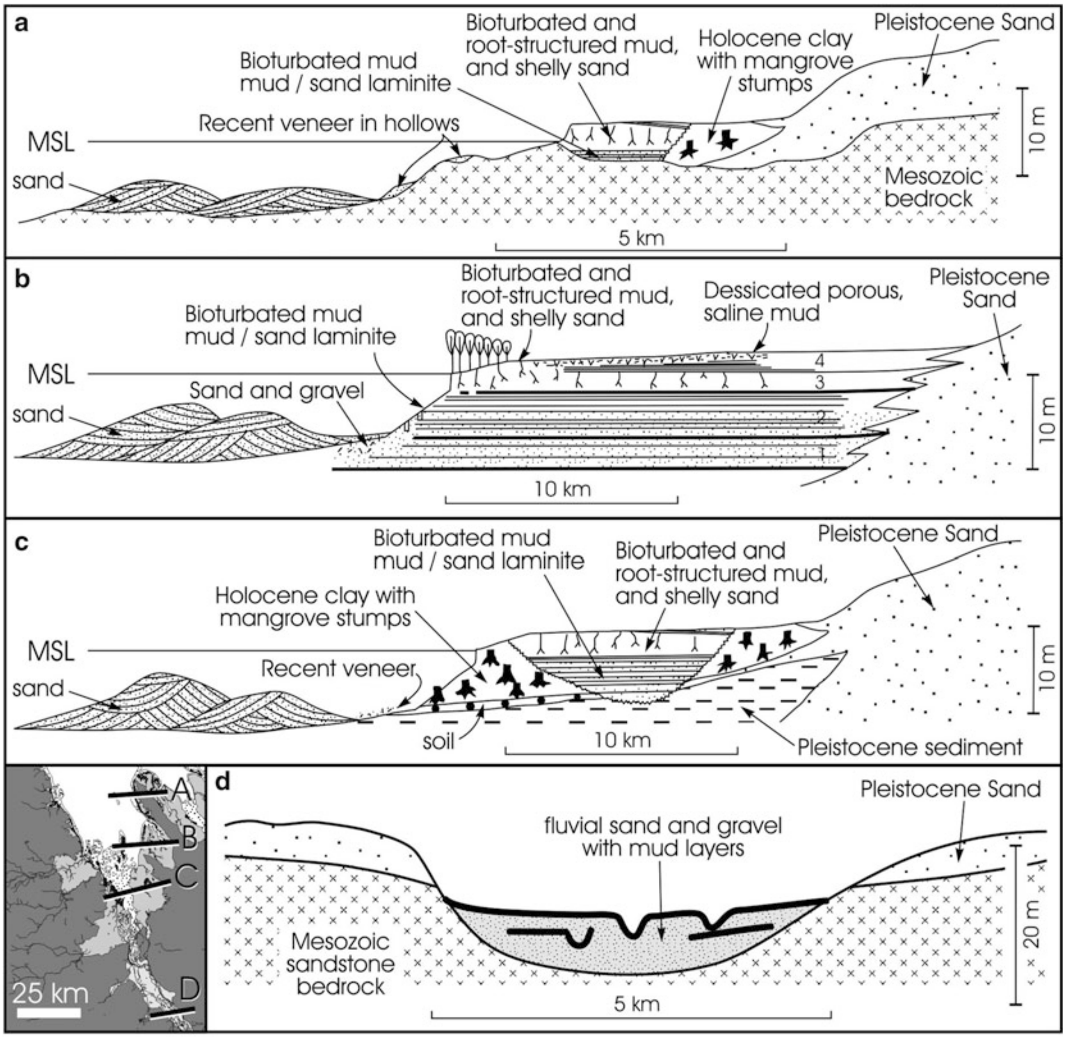
calcareous algae are present in estuaries of drier climates, carbonate mud may become a dominant or contributing component of the lithologic suite.

Large-scale whole-of-estuary stratigraphy

Large-scale stratigraphic accretion within an estuary is determined by the volume of sediment input. Using valley tracts (as the receiving basin reservoir) as examples for estuarine sedimentary fill, three types of sedimentary/stratigraphic sequences can be identified: (1) largely unfilled valley tracts and basins (wherein sedimentary sheets and veneers occur), (2) partly filled valley tracts and basins, and (3) fully filled valley tracts and basins. The most significantly developed stratigraphic sequences are developed in those estuaries that have large volumes of sediment delivered by fluvial and marine sources and a significant intra-estuarine sediment contribution. These comprise the partly filled and wholly filled valley tracts and basins.

Climate setting of the river drainage basin determines to a large extent the amount of weathering and erosion of source materials for the rivers and, hence, the volume and types of sediment and particle sizes that may be delivered to the estuary. Estuaries that have low influx of riverine sediment will be sedimentologically depauperate, and here the estuaries may consist of a rock-floored and rock-walled basins or may have only a veneer of modern sediment on the postglacial unconformity. Where there is little sediment delivery from marine environments, then, equally, the marine part of the estuary will also be sediment depauperate. In the sediment veneers, facies changes will occur reflecting the environments from river to sea, and the sediments of such estuaries will consist of a thin sheet of riverine sediment, grading to sediment with an estuarine signature, to that of marine sediment in a river-to-sea transect.

The stratigraphy of fully shoaled or nearly filled estuarine basin sequences has a common pattern: a pre-estuarine stratigraphy representing Pleistocene deposits, palaeosols,

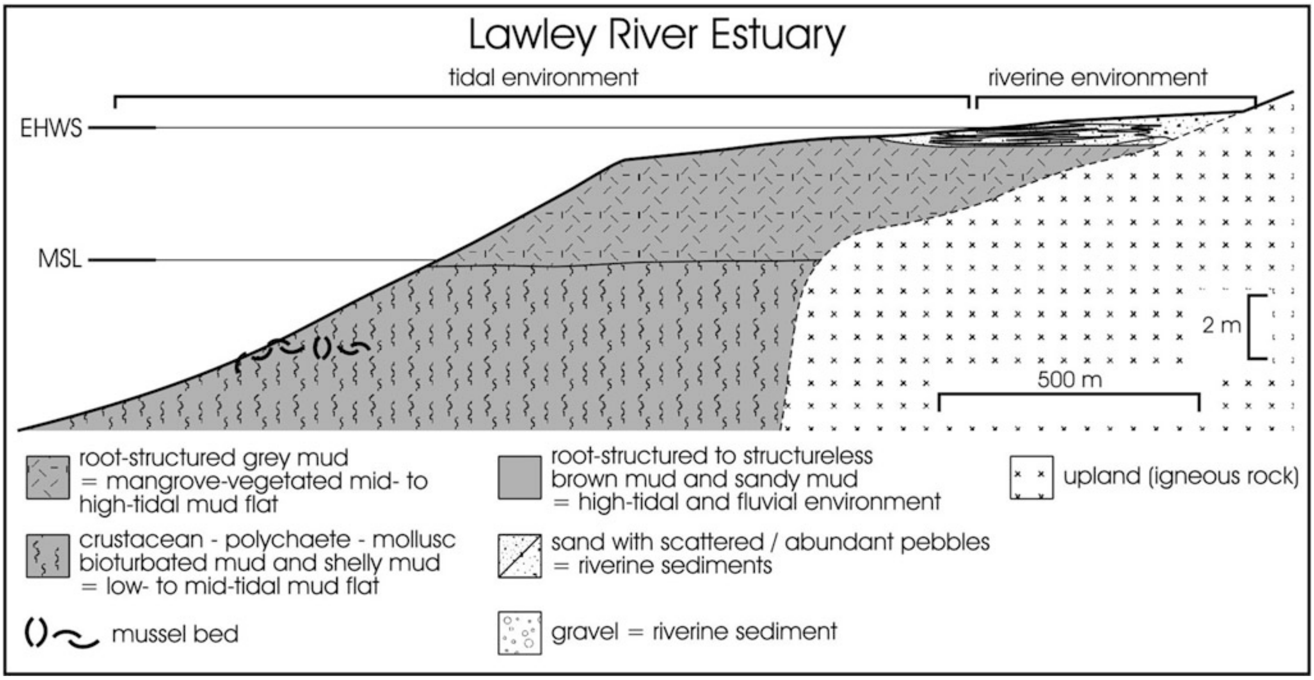


(adapted from Semeniuk 1980,1981, Semeniuk & Brocx 2011)

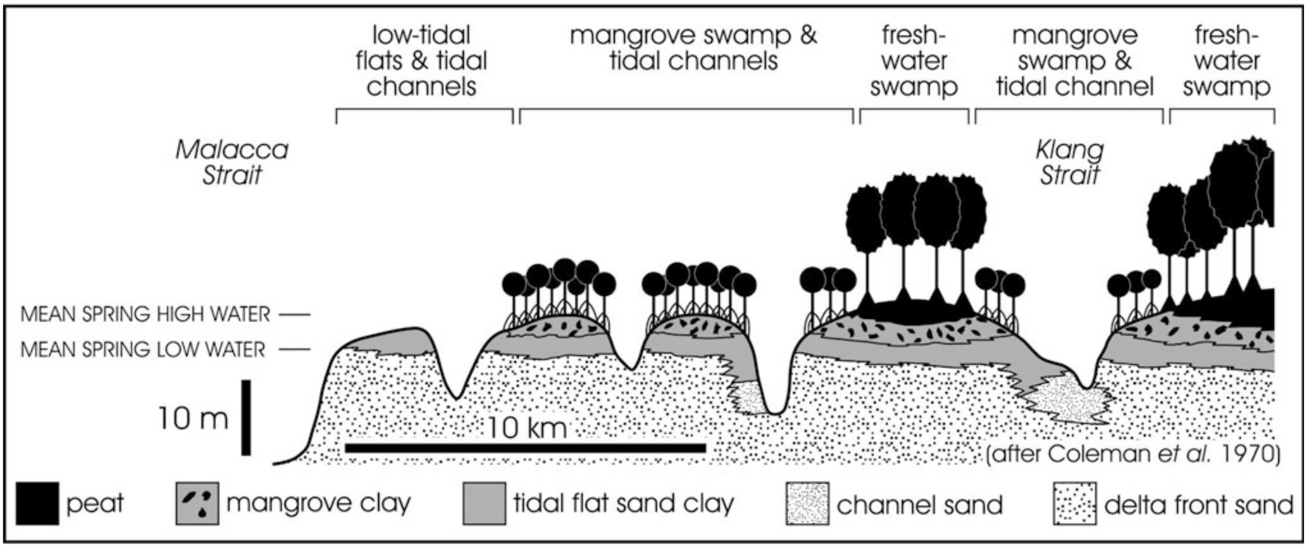
Stratigraphy of Estuaries, Figure 14 A series of transverse complex stratigraphic transects from south to north along the Fitzroy River Estuary. The transects show fluvial-dominated stratigraphy to the south (*inset D*), followed by complex relationship further north between early Holocene mangrove-developed mud deposits with large-scale cut and fill of later Holocene shoaling sand-to-mud tidal deposits (*inset C*), a shoaling Holocene sand-to-mud sequence in central parts (*inset B*), and veneers on pre-Holocene sediments and rock to the (distal) north (*inset A*) (Information from Semeniuk, 1980, Semeniuk, 1981 and Semeniuk and Brocx, 2011).

or bedrock, overlain by fluvial deposits, and then estuarine deposits, followed by either a capping of deltaic deposits, post-estuarine alluvial plain deposits, or transgressive barrier dune deposits.

At the scale of the estuarine basin, each of the estuarine sub-environments listed above (viz., the major estuarine-head delta(s); subsidiary deltas; marginal estuary alluvial fans; spits and their lagoons; marginal estuary environments, comprising platforms, tidal flats with some vegetated by mangrove and/or salt marsh, tidal flats with biostromes; central estuary basin, comprising deepwater environments; central estuary channels; tidal channels; tidal channels fringed by biostromes; central estuary shoals with some vegetated by mangrove and/or salt marsh; margin of leeward side of the barrier; flood-tidal delta; and washover fans) generates a specific and distinctive suite of sediments and, with accretion, will develop either a longitudinal sequence or a mosaic of stratigraphic units along and across the estuary basin, respectively, as will be shown later in the case studies. The extent that individual facies are dispersed, or migrate laterally or downslope, will determine the extent that they are recorded in adjoining facies and the extent that interfingering, interlayering, or encroachment takes place and hence the extent of the complexity of the whole-of-estuary stratigraphy.



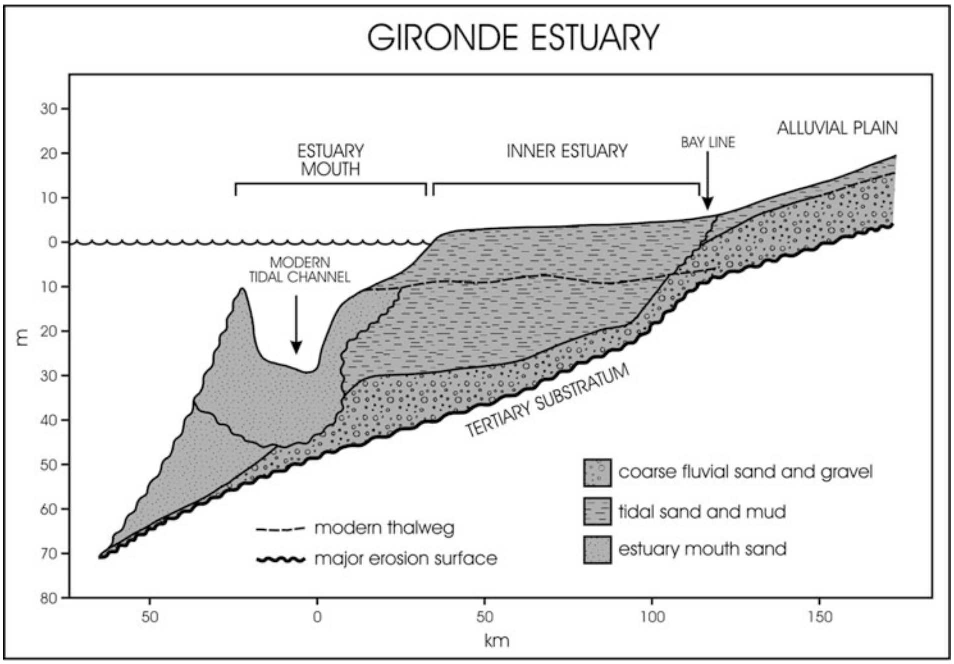
Stratigraphy of Estuaries, Figure 15 Longitudinal stratigraphic transect in the Lawley River Estuary showing a mud-dominated sequence. Fluvial deposits to the south are minimal on bedrock as compared to the thickness of the tidal and subtidal mud deposits (Information from Semeniuk, 1983, Semeniuk, 1985a).



Stratigraphy of Estuaries, Figure 16 Longitudinal stratigraphic transect in the Klang-Langat Estuary showing a sand-to-mud shoaling sequence, capped by freshwater peat (Modified from Coleman et al., 1970).

The expression of stratigraphy at the estuarine basin scale is a large-scale stratigraphy, which focuses on the geometry and interrelationships of the various gross facies within the estuary. For complex estuaries, the stratigraphy at the large scale will be an aggregate or amalgamation of juxtaposed and/or onlapping facies that have been formed at the small scale. Most studies of estuarine stratigraphy have been at the basin scale of a large-scale stratigraphy and, in particular, longitudinal stratigraphy (Dalrymple et al., 1992).

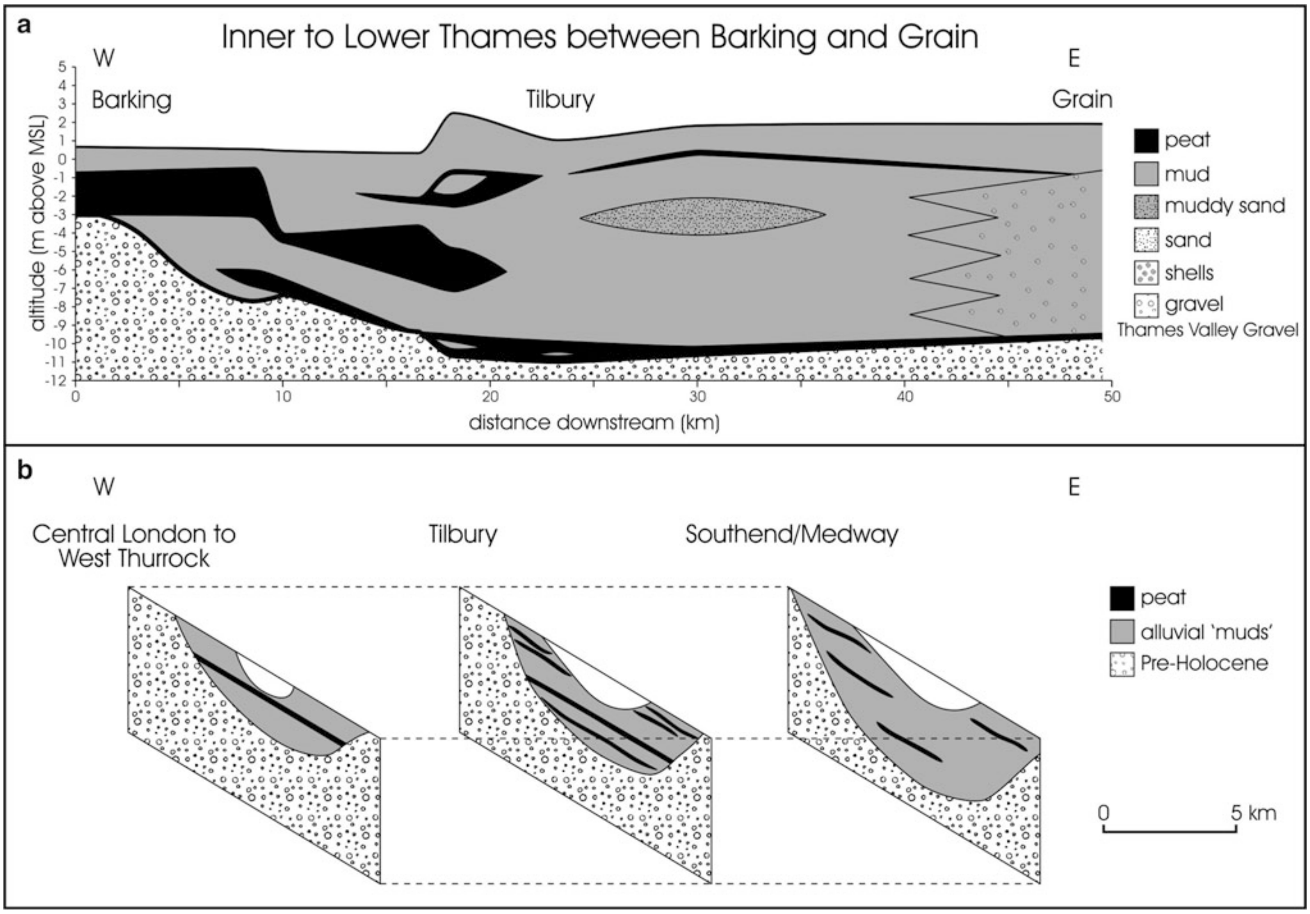
To date, there has been an emphasis on valley-fill estuarine stratigraphy (Dalrymple et al., 1992) usually within



Stratigraphy of Estuaries, Figure 17 Longitudinal stratigraphic transect in the estuary of the Gironde showing a sand-and-mud shoaling sequence and the relationship between the estuarine fill and the fluvial sediments (Modified from Allen and Posamentier, 1993).

a linear single valley tract, and this has provided valuable models of the evolution and architecture of whole-of-estuary stratigraphy as summarized in Figures 10, 11, 12, and 13. Figures 10 and 11 show generalized stratigraphy in a wave-dominated and a tide-dominated estuary, in a context of transgression, followed by sedimentary progradation, with simplified lithologic suites. Figures 12 and 13 show generalized stratigraphy in a wave-dominated and a tide-dominated estuary, in a context of the simplified lithofacies located according to the sub-environments within the estuary, but not as a progradational sequence.

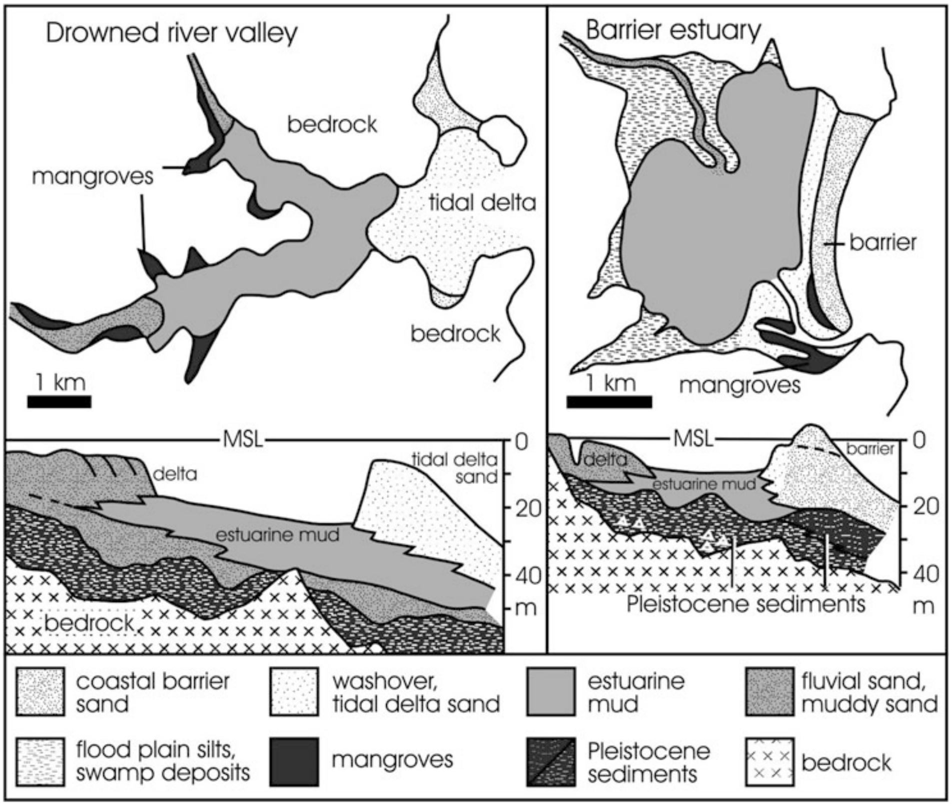
The stratigraphic models for estuaries, focused on wave-dominated types and tide-dominated types within linear, single valley tracts (Dalrymple et al., 1992; Figures 10, 11, 12, and 13), have provided an overarching view of their facies and stratigraphic architecture and have presented the stratigraphic evolution of an estuary in a conceptualized and uniform manner. However, while this overarching approach for incised valley systems provides a useful model to characterize many estuarine sedimentary fills and their longitudinal stratigraphy, it does not provide a framework for, or explanation of, the full range of relatively homogeneous stratigraphy or for the longitudinally and transversely heterogeneous stratigraphy found in many other types of estuaries in the variety of climatic settings and other coastal settings in which they occur. The South Alligator River Estuary of northern Australia, the Lawley River Estuary of northwestern Australia, and the Leschenault Inlet Estuary of southwestern Australia, among others, stand as examples of contrasting stratigraphic fills in estuaries. While the longitudinal facies variation in an incised valley estuary provides some degree of heterogeneity in such estuaries, the full range of heterogeneity, or even homogeneity, has not been addressed in these models. For instance, heterogeneity can be determined by deltas, i.e., on the number of deltas, their positions in the estuary and hence their stratigraphic contribution to the head and to the central parts of the estuarine embayment, their position relative to basin fetch, and hence whether they are wave-, tide-, or fluvially dominated, and the volume and type of sediment delivered by contributing river(s), and thus whether the deltas are mud-dominated, sand-and-mud-dominated, or sand-dominated. In the complex Peel-Harvey Estuary of southwestern Australia, for example, the deltas facing the prevailing summer southwesterly wind waves across a large estuary fetch are wave-dominated, while one in the south of the estuary is fluvially dominated (Semeniuk and Semeniuk, 1990a; Semeniuk and Semeniuk, 1990b). Similarly for the complex Walpole-Nornalup Inlet Estuary of southwestern Australia, each of the deltas faces estuarine waters of differing fetch and wind-wave trains such that they develop different delta types and delta stratigraphy.



Stratigraphy of Estuaries, Figure 18 Longitudinal stratigraphic transect in the estuary of the Thames showing a mud-dominated sequence, with local lenses of peat and sand (the stratigraphic section has been interpreted and constructed from information in Morgan, 2006, and Khan et al., 2011)

Estuarine stratigraphic patterns will also depend on hydrodynamic patterns, dispersal of sediments, estuarine geometry, and factors which can result in cross-estuarine heterogeneity or in facies asymmetry. For example, the Leschenault Inlet Estuary of southwestern Australia, a north–south-oriented shore-parallel estuarine lagoon, is subject to southwesterly wind waves that rework its eastern shores such that it is sand-dominated, whereas the western shore is spit, chenier, mud, and muddy sand-dominated (Semeniuk, 2000). The Peel-Harvey Estuary of southwestern Australia, mentioned above, provides another example. Although functioning ecologically and hydrochemically as a single estuary, it is geomorphically a compound estuary. It has developed heterogeneous stratigraphy as a result of residing in a complex geological framework, with local source materials, river dynamics, fetch, and estuarine hydrodynamic processes (Semeniuk and Semeniuk, 1990a). For instance, its shores facing a long fetch are wave-dominated and composed of prograded sandy beach ridges, while those sheltered from the prevailing wave directions are mud-dominated. Also, estuaries developed by marine flooding of a complex meandering river on a coastal plain will result in a geomorphically and sedimentologically complex estuary, with a resultant complex whole-of-estuary stratigraphy. In summary, the heterogeneous geometry, setting, and internal features of estuaries may directly contribute to a heterogeneous stratigraphy.

Thus, heterogeneity, complexity, and variability in an estuary, in terms of shape, hydrodynamics (viz., river vs. internal estuarine processes vs. magnitude of tides, ocean waves, and wind), and sediment sources and volumes, will result in complex estuarine topography, internal geomorphology, and facies. For instance, in shore-parallel elongate estuary, with a sand supply from eroding estuarine shores, and local sand (and not riverine sand), under appropriate wind directions and wind waves, there can be a major source of sediment transported alongshore

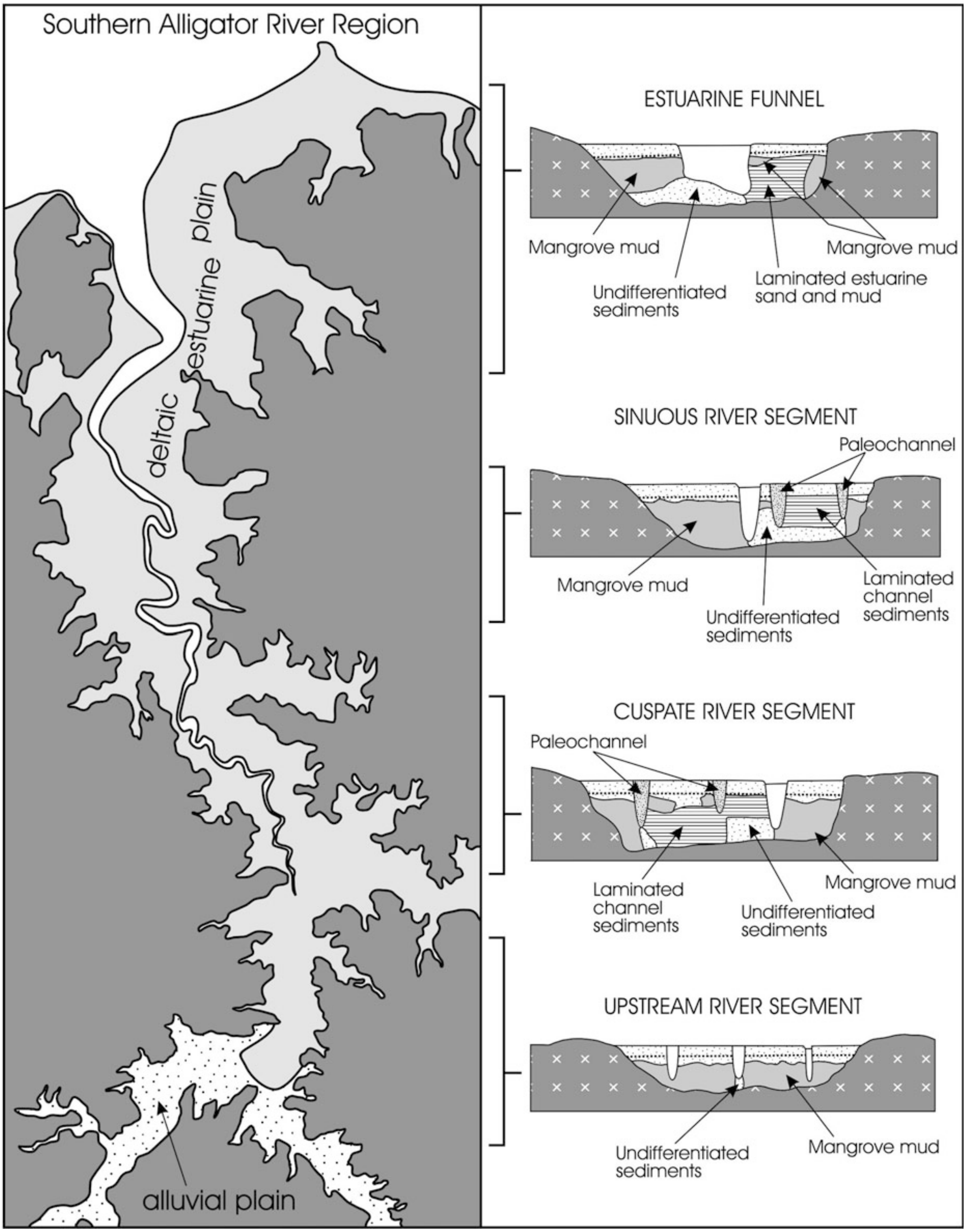


Stratigraphy of Estuaries, Figure 19 Generalized longitudinal stratigraphic transects in two estuary types from southeastern Australia showing the style of stratigraphic filling in these sand-and-mud-filled estuaries (Modified from Roy et al., 1980 and Roy, 1984).

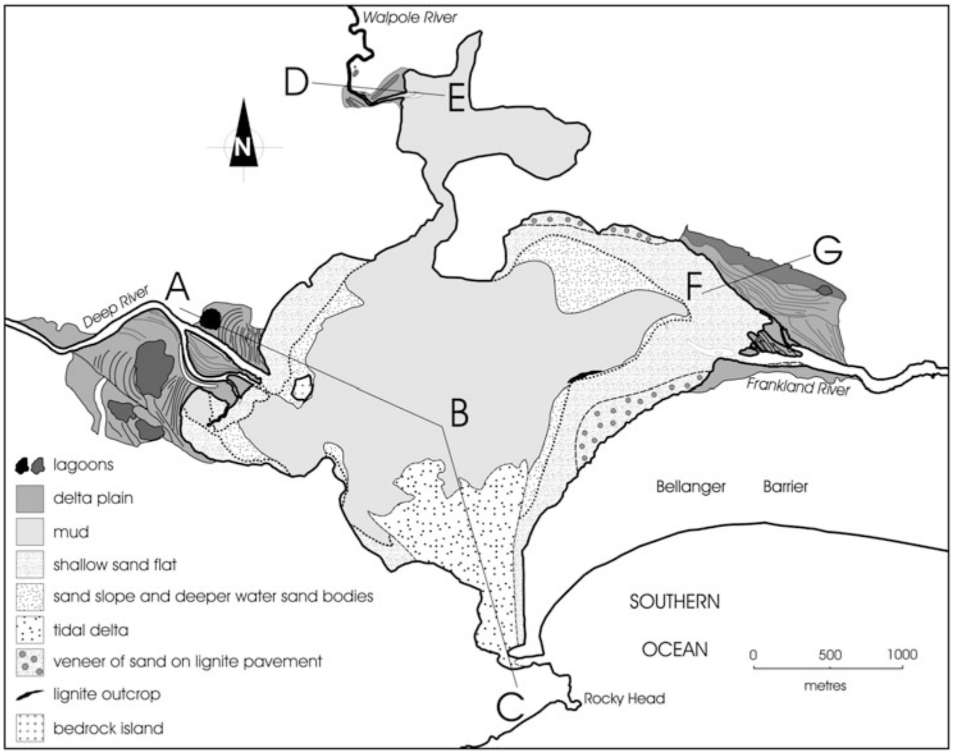
to form beach ridges and cuspate forelands, that can prograde into the estuarine basin. Similarly, a shore-parallel elongate estuary with a barrier dune migrating inland may transgress into the estuary or partition the estuary or at the least contribute to estuarine shore facies. Also, elongate estuaries with mobile sand along their shore (derived from an eroding shore mentioned earlier) can develop accretionary spits and cuspate forelands that segment the estuary (Zenkovitch, 1959; Bird, 1969), with the accretionary sedimentary bodies, manifesting as sand bodies "invading" from the estuary margins into the interior of the estuary, locally encroaching over basinal muddy deposits. In contrast, an estuary founded on a bifurcating tributary system, but now largely barred by a barrier, may be digitate/palmate in geometry, with complicated hydrodynamics resulting in complicated sedimentary dynamics and facies (of sandy spits, bars, cuspate forelands, and mud-filled lagoons leeward of these spits, bars, and cuspate forelands), as the wind-wave fields interact with complex estuarine form and shore orientations.

Complexity in stratigraphy of estuarine valley fills can also reflect the various depositional regimes in relation to sea-level lowstands, transgressive phases, and sea-level highstands. Allen and Posamentier (1993), for instance, document the stratigraphic sequence of an incised valley fill in the Gironde Estuary, France, recording a diverse assemblage of lithofacies (that are grouped into lowstand, transgressive, and highstand system tracts) and sedimentation within the current estuary (in terms of estuary mouth, estuary funnel, a zone of varying types of estuarine channels, and the alluvial plain). The valley-fill stratigraphic sequence begins during the Würm global sea-level lowstand, with accumulation of a continuous unit of relatively thin fluvial gravel and coarse sand in the thalweg of the incised valley. The transgressive system tract, accumulated during the Holocene sea-level rise, comprises the bulk of the incised valley fill and forms a landward-thinning wedge of tidal-estuarine sands and muds. In the estuary mouth, these are overlain by a thick unit of coarse-grained tidal-inlet and tidal-delta sands. The highstand system tract forms a seaward-prograding, tide-dominated estuarine bayhead delta that has been gradually filling the estuary since the post-Holocene stillstand.

In addition, complexity and variability can result from climate effects and climate setting, which can determine factors such as the formation of peat, the extent of



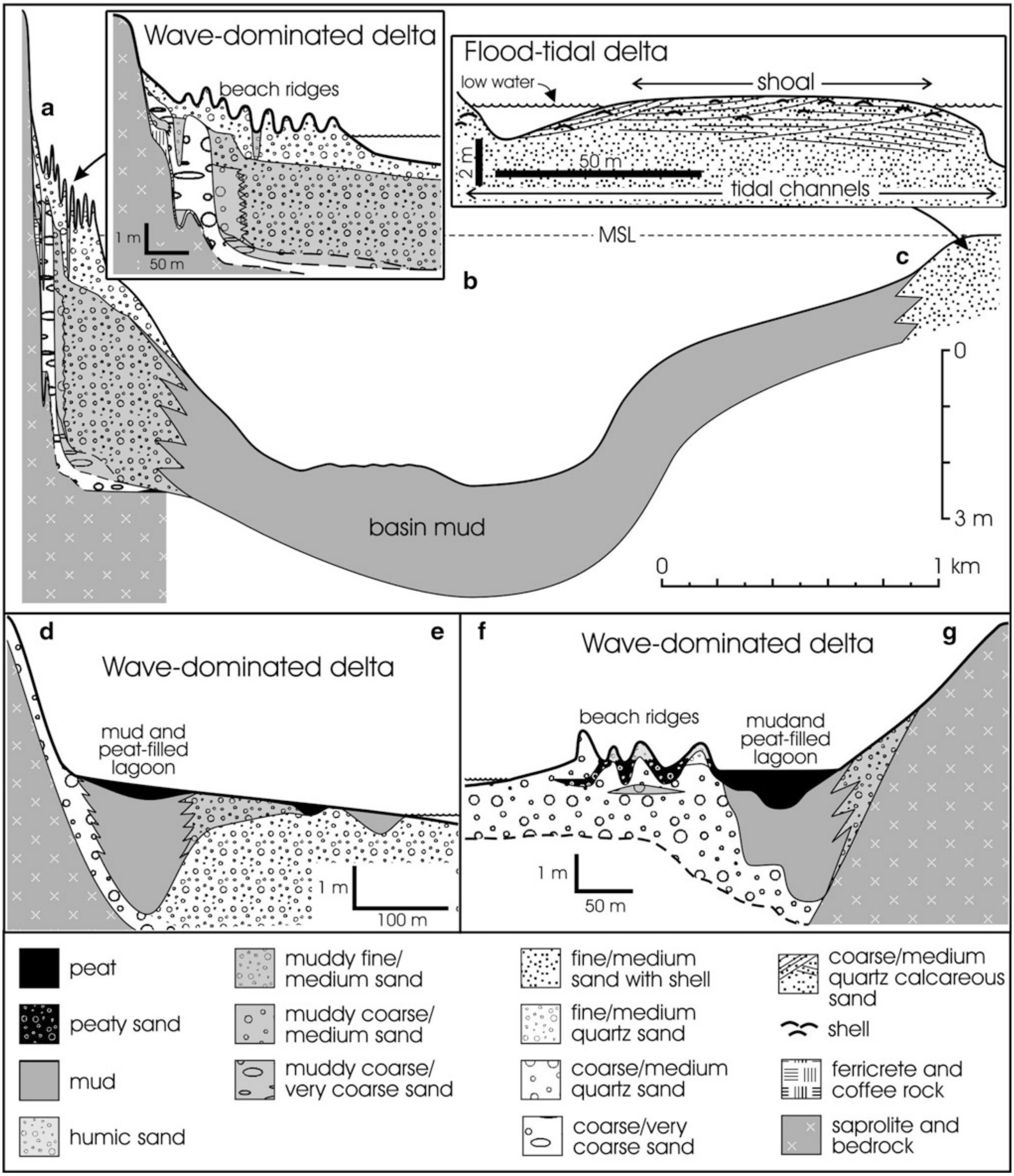
Stratigraphy of Estuaries, Figure 20 Transverse stratigraphic transects located along the length of the South Alligator River valley tract in northern Australia showing a sand-and-mud-filled system and the stratigraphic architecture along different segments of the estuary (Modified from Woodroffe et al., 1985, Woodroffe et al., 1986).



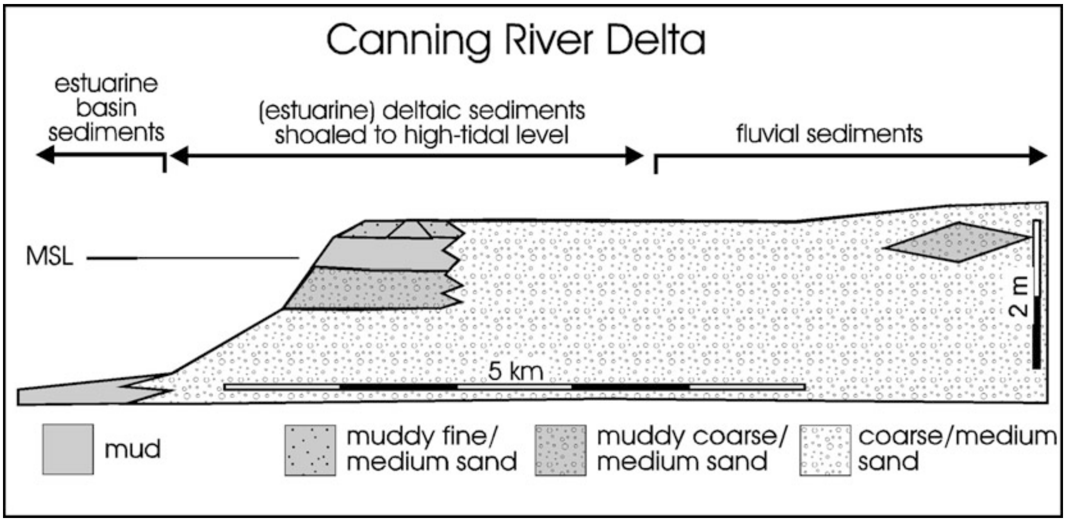
Stratigraphy of Estuaries, Figure 21 Location of stratigraphic transects in the twin ria Walpole-Nornalup Inlet Estuary. The map of the estuary also shows the location of the three deltas that enter the estuary, the shallow water sand platforms, the flood-tidal sandy delta, and the deepwater mud basin (Information from Semeniuk et al., 2011).

freshwater seepage from uplands bordering the estuarine shore, the biodiversity and productivity of plant and animal biota, and the processes and products of evaporation. Biogeography and animal productivity can determine the extent that biostromes and reefs are developed as stratigraphic units. Plant productivity can determine the extent that peat in high-tidal and supratidal areas and seagrass-trapped sand sheets in tidal and shallow subtidal areas are developed as stratigraphic units. A selection of examples of the variability of large-scale whole-of-estuary stratigraphy from a range of case studies is provided below. The examples for the large-scale whole-of-estuary stratigraphy are as follows (the shape and size of most of these estuaries are shown in Figure 1):

1. Valley-fill stratigraphy in macrotidal settings in a tropical climate using the relatively geomorphically simple but sedimentologically complex tide-dominated sand-and-mud-filled Fitzroy River Estuary (King Sound) and the relatively geomorphically and sedimentologically simple tide-dominated mud-filled Lawley River Estuary of northwestern Australia (Semeniuk, 1980, 1981, 1983, 1985a; Semeniuk and Brocx, 2011; Figures 14 and 15)
2. Valley-fill and gulf-fill stratigraphy in a macrotidal setting in a tropical climate using the relatively geomorphically simple tide-dominated sand-and-mud-filled Klang-Langat Estuary in Malaysia (Coleman et al., 1970; Figure 16)
3. Valley-fill stratigraphy in a macrotidal setting in a temperate climate using the relatively geomorphically simple tide-dominated estuary of sand-and-mud-filled Gironde and the tide-dominated sand-and-mud-filled Thames estuary (Allen and Posamentier, 1993; Morgan, 2006; Khan et al., 2011)
4. Valley-fill stratigraphy in a microtidal to mesotidal setting in a temperate climate using the moderately geomorphically complex mixed wave and tide-dominated sand-and-mud-filled estuaries of southeastern Australia (Roy et al., 1980; Roy, 1984)
5. Stratigraphy in a meandering valley, on an alluvial plain in a macrotidal setting in a tropical climate using the relatively geomorphically complex tide-dominated sand-and-mud-filled South Alligator River Estuary of northern Australia (Woodroffe et al., 1985, 1986)
6. Valley-fill stratigraphy in a microtidal setting in a temperate climate along a wave-dominated coast using the relatively geomorphically complex intra-estuarine wave-dominated sand-and-mud-filled Walpole-Nornalup Inlet Estuary (Semeniuk et al., 2011)
7. Stratigraphy in a meandering valley on a coastal plain in a microtidal setting in a temperate climate using



Stratigraphy of Estuaries, Figure 22 Longitudinal stratigraphic profile along the southern basin of the Walpole-Nornalup Inlet Estuary from a large delta to across the mud basin to the flood-tidal delta and two local transects across the other two deltas with their leeward mud-and-peat-filled lagoons. Insets show more stratigraphic detail of the sandy deltas and flood-tidal delta of the main transect. Information from Semeniuk et al. (2011). Note that two of the deltas are capped by peat or have peat in their stratigraphic development.



Stratigraphy of Estuaries, Figure 23 Longitudinal stratigraphic profile along the Canning River delta in the Swan-Canning Estuary of southwestern Australia showing a sand-and-mud fluvial system, a sand-dominated delta, and a mud-dominated estuarine basin.

the relatively geomorphically simple river-dominated sand-and-mud-filled Swan-Canning Estuary of southwestern Australia

8. Stratigraphy in a coastal estuarine lagoon in a microtidal setting in a subtropical climate using the moderately geomorphically complex river-dominated to tide-dominated sand-and-mud-filled Leschenault Inlet Estuary of southwestern Australia (Semeniuk, 2000)
9. Stratigraphy in a compound estuary on a coastal plain in a microtidal setting in a subtropical climate using the relatively geomorphically complex river-dominated to wave-dominated sand-and-mud-filled Peel-Harvey Estuary of southwestern Australia (Brown et al., 1980; Semeniuk and Semeniuk, 1990a; Semeniuk and Semeniuk, 1990b)

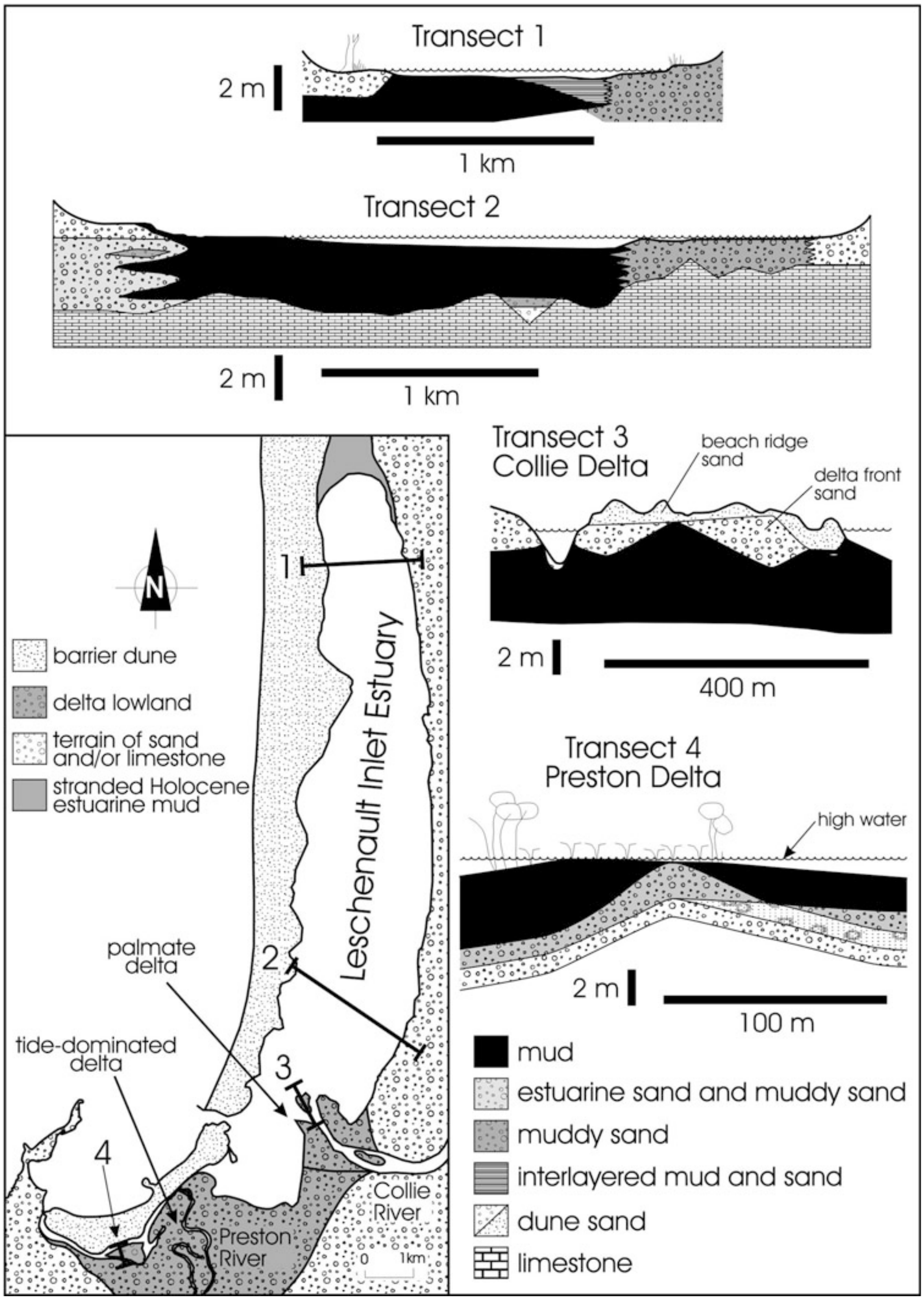
The examples span climates from tropical to temperate, tidal regimes from macrotidal to microtidal, and sedimentologically from sand-and-mud accumulations to mud-dominated systems.

Of the range of examples provided above, the Gironde Estuary, those of southeastern Australia, and perhaps the South Alligator River conform with the idealized model of incised valley estuarine stratigraphy. The other estuaries provide variations and complications as follows: (1) they are mud-dominated or mud-dominated with some peat (Lawley River Estuary and the Thames, respectively); (2) they contain earlier Holocene deposits accumulated in a different climate and different height of MSL (Fitzroy River Estuary) and, as such, the stratigraphic fill is not simple; (3) they are capped by peat because of their humid climate setting (Walpole-Nornalup Inlet Estuary and Klang-Langat Estuary); (4) they are dominantly a mud-filled estuarine lagoon with sand marginal platforms and with a small component of deltaic deposits, and the estuarine fill will be a river-derived mud ribbon capped by estuarine sand sheets reworked from barrier dunes and Pleistocene dunes and finally covered by sands of the retreating coastal barrier dunes (the Leschenault Inlet Estuary); (5) they comprise small component of deltaic deposits in essentially a sediment-depauperate estuarine system (the Swan-Canning Estuary); and (6) they comprise complex compound systems with a variety of shoals and shoreline protuberances that complicate the idealized transition from delta to estuarine mouth shoals (the Peel-Harvey Estuary) (Figures 17, 18, 19, 20, 21, 22, 23, 24, 25, and 26).

Summary and discussion

Stratigraphy of an estuary can be complex and dependent on a wide range of variables, from the large scale involving regional setting, estuary origin and shape, and oceanography to the small scale involving lithogenesis and biota. While there is complexity in estuaries and much variability between them, for each of the estuarine types, with their own hydrodynamic settings and sediment sources, there appears to be a recurring pattern of (a limited range of) stratigraphic fills both at the small scale of the environment-specific level and at the large scale of the whole-of-estuary level.

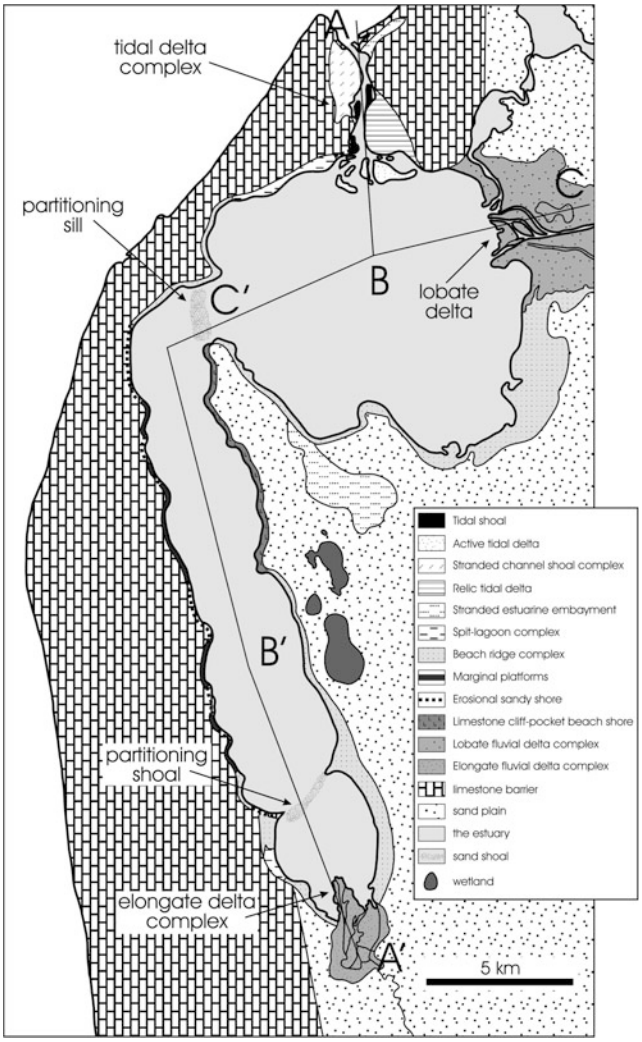
This contribution provided information on stratigraphy of estuaries at two scales – that of the facies-specific scale and that of the whole-of-estuary scale in longitudinal and



Stratigraphy of Estuaries, Figure 24 Transverse stratigraphic transects located along the length of the Leschenault Inlet Estuary showing a sand-and-mud-filled estuary and sand-and-mud deltas. Information from Semeniuk (2000). The mud fills the central elongate estuarine basin, and sand underlies the shallow water platforms.

transverse profiles, in relatively stratigraphically homogeneous estuaries, and in heterogeneous estuaries.

A major factor in determining the lithology, and both the small-scale facies-specific stratigraphy and the large-scale whole-of-estuary stratigraphy, is the climate setting of the estuary. Climate can result in a range of lithologic types and stratigraphic types, particularly in the tidal and supratidal environments, and it will



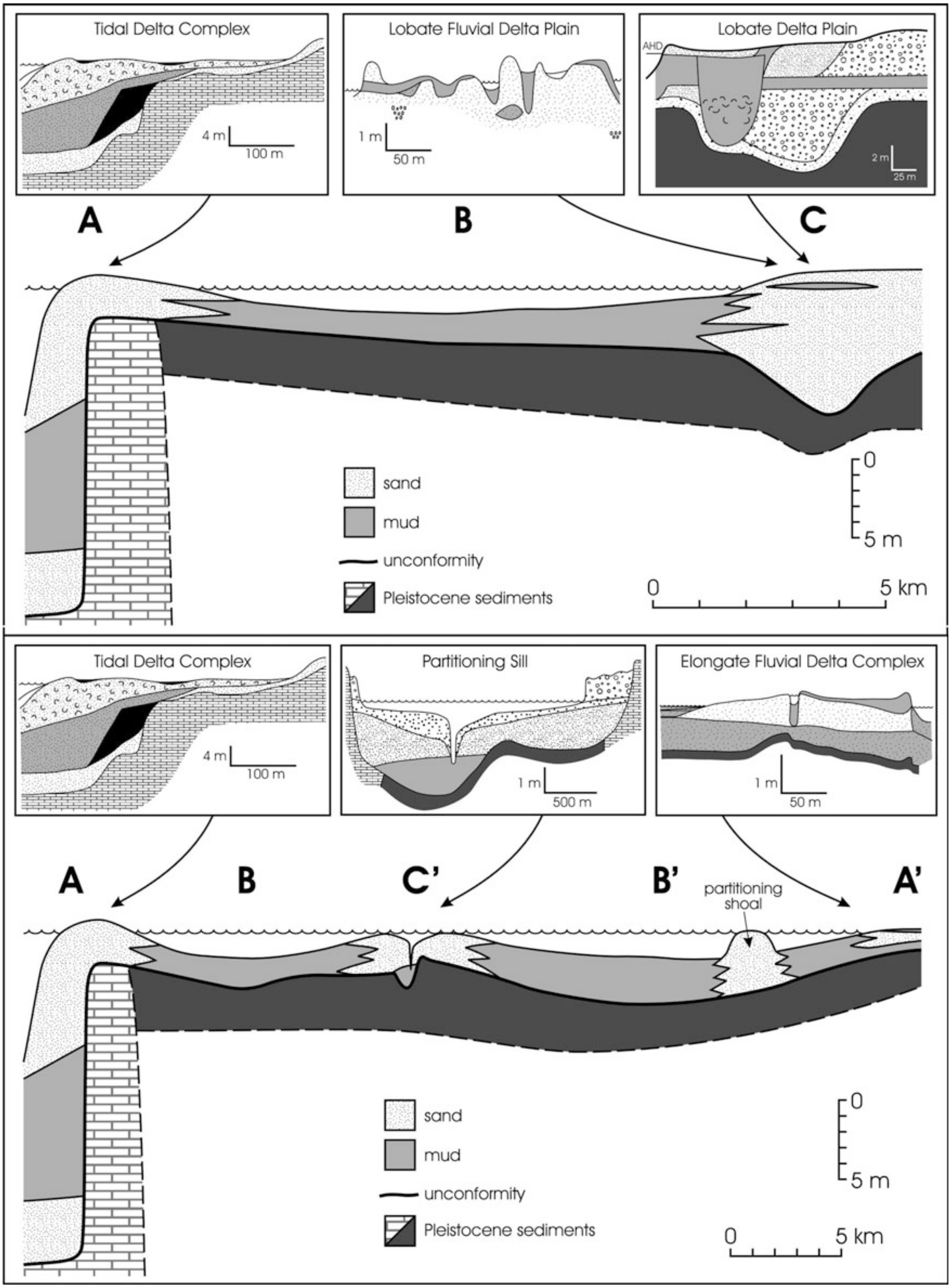
Stratigraphy of Estuaries, Figure 25 Location of stratigraphic transects in the complex compound Peel-Harvey Estuary. The map of the estuary also shows location of the three deltas that enter the estuary, the shallow water partitioning shoals, the flood-tidal sandy delta, the deepwater mud basin, and the complexity of shore types (Modified from Semeniuk and Semeniuk, 1990a; Semeniuk and Semeniuk, 1990b).

determine the extent that fluvial sediment will contribute to the filling of an estuarine basin. It will also have a particular effect on the sediments and stratigraphy of facies that have become emergent (shoaled) and inhabited by tidal and supratidal vegetation. The shoaled stratigraphy of estuaries in tropical humid climates, for instance, may be capped by mangrove-vegetated mud, or at least have a heavy imprint of vegetation bioturbation in tidal/supratidal environments, and also contain a rich diversity of shell benthos in lower tidal and shallow subtidal environments. The shoaled stratigraphy of estuaries in temperate humid climates or tropical humid climates may be capped by peat, or by sand or mud, with a heavy imprint of vegetation bioturbation in tidal/supratidal environments, and contain a relatively depauperate diversity of shell benthos in lower tidal and shallow subtidal environments. The stratigraphy of estuaries in drier climates may have less peripheral vegetation and tend to preserve sedimentary lamination. Also, if calcareous algae are present in estuaries of drier climates, carbonate mud may become a component of the lithologic suite.

Another major factor in determining the lithology, the small-scale stratigraphy, and the large-scale stratigraphy of estuaries is the coastal hydrodynamic setting of an estuary and the intra-estuarine hydrodynamics, i.e., whether the estuary is river-dominated, tide-dominated, or set along a wave-dominated coast. Internal to an estuary, the hydrodynamics determined by how far tides penetrate into the estuary, and the wind patterns and fetch particularly influence the array of estuarine facies that are developed.

For incised valley systems, with an up-valley river source, the stratigraphy is longitudinally graded because of down-valley facies of estuaries, and this will be a major factor in the development of the stratigraphic architecture of an estuary. The nuances of ocean-side facies development in the down-valley facies transition will be determined by whether the oceanographic setting is wave-dominated, tide-dominated, or mixed wave- and tide-influenced. In wave-dominated environments, the ocean-side environments will often develop a barrier, which under strong wind conditions or coastal retreat conditions may transgress over the estuarine deposits (Semeniuk 1985b). The geomorphic perforations in the barrier are sites of flood-tidal deltas, with their specific stratigraphic geometry and lithologic signature. During storms, the barrier may be breached or overtopped to form washover sediment fans or lobes that encroach into the estuary prograding over deeper water lithofacies. In tide-dominated environments, the ocean-side facies will often develop tidally oriented shoaling sand bars. But while there may be a recurring overall pattern to facies/stratigraphic architecture longitudinally along the whole of estuary in both situations, the details of lithology in site-specific environments are determined or influenced by the local effects of climate and by the biodiversity and abundance of the biota.

Outside of incised valley systems, the stratigraphy of estuaries will be determined by the complexity of the array and mosaics of environments within the estuary as a consequence, in part, of the type of estuary, e.g., shore-parallel estuarine lagoon, digitate/palmate estuarine lagoon, meandering estuary on a coastal plain, or compound estuary. The sedimentary facies in each of these estuarine types will be determined by different types of intra-estuarine processes such as wind-wave development in relation to fetch, the depth of water, the extent of shoaling, climate influences, and biota. These types of estuaries provide the largest range of variability of stratigraphic fills and architectural style of stratigraphic fills.



Stratigraphy of Estuaries, Figure 26 Longitudinal stratigraphic profile from the north to the south of the Peel-Harvey Estuary: from the flood-tidal delta to across the mud basin, across the partitioning shoals, and to southern elongate digitate delta. The insets show details of the two northern lobate deltas, the elongate digitate delta, the stratigraphically complex flood-tidal delta, and the partitioning sandy sill (Information from Brown et al., 1980 and Semeniuk and Semeniuk, 1990a; Semeniuk and Semeniuk, 1990b).

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Cross-references

Biogenous Sediment
Estuarine Sediment Composition
Sediment Sorting
Sediment Transport
Washover Fans
Washovers

STRUCTURALLY DOMINATED ESTUARY

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Synonyms

Structurally built estuary; Tectonically produced estuary

Definition

A structurally dominated estuary is one where the shape of the estuarine basin is primarily determined by the long-term geological history of the coast.

Description

Two types of structural estuaries have been identified: (1) where the tectonic setting has determined the nature of the estuarine basin and (2) where the sediments within the estuary have been impacted by neotectonics after their deposition (Quivira, 1995). Structural estuaries most commonly occur on tectonically active coasts where faulted landscapes produce basins which can be flooded by the sea. Structural grabens, eroded volcanic calderas, and uplifted river valleys are all examples of systems that can be considered to be structurally influenced (Hume and Herdendorf, 1988; Hume, 2003). In highly jointed bedrock, ria-type estuaries form, and these can also be classified as structural estuaries (Kennedy, 2011). The infill of structurally dominated estuaries is complex, being driven by sediment supply, accommodation space, and process dominance (Kennedy, 2011), with those sequences on tectonically active coasts preserving multiple transgressive and regressive sequences in response to vertical land movement (Wilson et al., 2007).

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Cross-references

Emergent Shoreline
Submerged Coasts
Submergent Shoreline
Tectonic Eustasy
Uplifted Coasts

SUBAQUEOUS SOILS

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Definition

The foundation of every estuarine ecosystem is the water column and underlying substrate that support the plants and animals living in these unique habitats. In most cases, the substrate is considered sediment, and the materials are classified based simply on their grain size (i.e., mud, sand). In the late 1990s, pedologists began developing a soil science approach to study, map, and classify shallow subtidal substrates as soils (http://nesoil.com/sas/index.htm). These studies recognized that shallow subtidal substrate often supports rooted plants (submerged aquatic vegetation; SAV) and that a range of physical, chemical, and biological processes operate in these systems resulting in the formation of subaqueous soils. Important processes in subaqueous soils include accumulation of organic matter, accumulation of sulfides (sulfidization), sedimentation, and bioturbation (soil mixing). Early studies of subaqueous soils focused on methodologies to sample and characterize these soil materials. Models were developed to link shallow subtidal landscapes (i.e., flood-tidal deltas, washover fans, submerged beaches) to specific soil types. Understanding the processes, characteristics, and subaqueous soil-landscape relationships aided in developing soil taxa to classify the soils which could be used to map the subtidal components of estuaries. These taxa are incorporated into the Entisol and Histosol orders of the US national soil classification system known as Soil Taxonomy (ftp://ftp-fc.sc.egov.usda.gov/NSSC/Soil_Taxonomy/keys/ebook/Keys_to_Soil_Taxonomy_11th_Edition.pdf).

More recent studies have focused on the application of subaqueous soils information to estuarine management issues such as shellfish and SAV restoration, identifying productive areas for shellfish aquaculture, carbon accounting, and determining which areas can be dredged and the materials placed on the upland (Rabenhorst and Stolt, 2012). The authors of the US National Coastal and Marine Ecological Classification Standard (CMECS; http://www.csc.noaa.gov/digitalcoast/publications/cmecs) recognized the value of its use and management interpretations and recommended that the subaqueous soils approach be employed to classify shallow subtidal substrates for use and management purposes. Mapping subaqueous soils is usually done in waters less than 5 m deep. Although the methods to create a subaqueous soils map are well developed, mapping subaqueous soils is still in its infancy. Maps of selected estuaries are available on the Atlantic coast of the USA (in Maine, Rhode Island, Connecticut, New York, New Jersey, Delaware, Maryland, and Florida).

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Cross-references

Sediment Grain Size

SUBLITTORAL ZONE

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Synonyms

Subtidal zone

Definition

In an estuary, sublittoral refers to the zone immediately below the eulittoral zone extending outward from the neap low tide mark at the shoreline. In an ocean, sublittoral refers to the zone extending from the neap low tide mark at the shoreline to the outer edge of the continental shelf.

Description

Sublittoral habitats in an estuary lie within the photic zone and thus are typically highly productive. Numerous flora and fauna inhabit the sublittoral zone forming rich biotic communities (Day et al., 2012; Levinton, 2013). This is a subaqueous environment susceptible to the vagaries of physical and chemical factors, resulting in variable spatial and temporal biotic responses (Allaby and Allaby, 1999; Kennish, 2001).

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Cross-references

Benthic Ecology
Littoral Zone
Shoreline

SUBMERGED COASTS

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Synonyms

Submergent coasts

Definition

Submerged coasts are defined as coasts formed by the relative submergence of a landmass via sea-level rise and/or by land subsidence or by both factors.

Description

Most early coastal classification schemes are based on the observation that coastal landforms are largely the product of sea-level variations. Such classifications distinguish between emerged and submerged coasts (Johnson, 1919). More sophisticated analyses have attempted to establish coastal classifications by balancing two sets of factors in the coastal configuration, one historical and one contemporary. In the first, the coastal type is controlled by plate tectonics resulting in emergence-submergence categories. Historic factors must be assessed such as the nature of the sea-level change over the last 10,000 years of the postglacial transgression. Additionally variations in wave energy, tidal regime, and biogeographic agents and processes must be considered. Valentin (1952) incorporated these factors in an elegant diagram that combined advancing and retreating coasts with emerging and submerging ones. Most of the submerging coasts correlate with coastal retreat or land erosion. The main shortcoming of these early classifications is the emphasis on geological control that places less significance on the role of hydrodynamic processes.

The coupling mechanisms in coastal submersion are sea-level rise and land subsidence. Between 20 and 6 kyr ago, driven by the melting of Northern Hemisphere ice caps, global sea level increased ~120 m. This rise in sea level caused the rapid submergence of vast areas of the continental shelf. In the last 6 kyr, global sea level has remained more or less stable during an interglacial period (Pirazzoli, 1996). Despite being more variable in time and space, land subsidence may result from different tectonic and neotectonic processes or by the accumulation and compaction of sediments. During the twentieth century, many coastal areas experienced human-induced land subsidence caused by groundwater withdrawal or oil and gas extraction, among others. Regulation of river flow and the construction of breakwaters that cut longshore sediment supply have also contributed to destruction and sinking of deltas (Bird, 1993). Climate models, which take into account increasing greenhouse effects, estimate that the global temperature may increase 2 °C and sea level may increase 0.11–0.77 m over the next century, which will cause significant coastal flooding and submergence (Church et al., 2001).

Typical submerged coasts are drowned river and glacial valleys, often referred as *rias*, *calas*, *fiords*, *fiards*, *sharms*, and *sebkhas*. Although many of them can fit within a broad morphological classification of estuaries, from a hydrodynamic point of view, drowned valleys are not necessarily estuaries (Bird, 2008). Inlets formed by partial submergence of river valleys are known as *rias*, whereas *fiords* are steep-sided inlets at the mouth of valleys that were formerly glaciated. Inlets generated by Holocene marine submergence of formerly glaciated valleys and depressions in low-lying rock landforms are known as *fiards*. These steep-sided valleys, incised in limestone plateaus during low sea-level stages that have been submerged during the Holocene, are known as calas or calanques (Gómez-Pujol et al., 2013). Finally, in arid environments, such as those of the Red Sea and the Arabian Gulf, wadis developed during the Holocene marine transgression, leading to the formation of long, narrow marine inlets termed *sharms*. When these valleys exhibit a branched embayment, they are known as *sebkhas* (Castaing and Guilcher, 1995).

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Cross-references

Deltas
Mean Sea Level
Shoreline Changes

SUBMERGENT SHORELINE

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Synonyms

Submarine shoreline; Submerged coasts; Submergence; Subsided shoreline

Definition

A submergent shoreline is one that becomes flooded due to coastal subsidence or sea-level rise.

The Dutch coast is well known for its long-term crustal subsidence. This has given rise to a sequence of submarine shorelines and a transgressing sea (Jelgersma, 1961). Sediment loading in delta areas can induce subsidence (sedimento-isostasy) and the development of a submerging coast with a succession of submarine shorelines, drowned forests, and flooded coastal zones. Eustatic sea-level rise after the last Ice Age has been ~120 m, which has caused most coasts of the world to experience substantial inundation, leaving former shorelines and land surfaces in submarine positions (e.g., Daly, 1934).

Water extraction along coasts may also lead to human-induced subsidence, often resulting in local increases of relative sea level. In combination with subsidence of a delta environment, this may cause severe coastal problems, as in Bangkok.

Many old-tide gauges were installed in areas of heavy harbor construction, and the added weight of the construction can cause local compaction of the sediments and associated subsidence. NOAA (2013) has 204 tide gauge stations scattered all over the globe where the mean rate of sea-level rise is +0.75 mm/year.

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Cross-references

Climate Change
Mean Sea Level
Shoreline Changes
Submerged Coasts

SUSTAINABLE USE

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Synonyms

Sustainable Resource Use

Definition

Sustainable use is the use of resources in a way and at a rate that does not lead to long-term decline, thereby maintaining its potential to meet the needs and aspirations of present and future generations (adapted from UN, 1992).

Introduction

Natural resources are vital to the survival and development of the human population and to the world economy. However, the way in which natural resources, both renewable and nonrenewable, are used and the speed at which renewable resources are being exploited are rapidly eroding the planet's capacity to regenerate the resources and environmental services on which our prosperity and growth is based (CEC, 2005). According to the Millennium Ecosystem Assessment report, over the past 50 years, humans have changed ecosystems more rapidly and extensively than in any comparable period of time in human history, largely to meet rapidly growing demands for, namely, food and freshwater (MEA, 2005). If current patterns of resource use are maintained, environmental degradation and depletion of natural resources will continue. The issue has a global dimension. If the world as a whole followed traditional patterns of consumption, it is estimated that global resource use would quadruple within 20 years (CEC, 2005; EC, 2009). The negative impact on the environment would be substantial. The alternative can be to adopt a coordinated approach, anticipating the need to shift to more sustainable-use patterns, which can result in environmental and economic benefits at a global scale (CEC, 2005).

Sustainable use: the concept

In 1992, the Earth Summit was held in Rio de Janeiro (Brazil), with objectives built upon the hopes and achievements of Our Common Future Report (also known as the Brundtland Report), in order to respond to pressing global environmental problems (UN, 1987). Among other agreements, the United Nations Convention on Biological Diversity was adopted (UN, 1992). This convention, considered a key global instrument on conservation (Höft, 2008), establishes three main goals: (1) the conservation of biological diversity, (2) the sustainable use of its components, and (3) the fair and equitable sharing of the benefits from the use of genetic resources. The United Nations Convention on Biological Diversity conveys to decision-makers that natural resources are not infinite. Thus, it sets out a new philosophy for the twenty-first century: sustainable use (SCBD, 2000). While past conservation efforts aimed at protecting particular species and habitats, the convention recognizes that ecosystems must be used for the benefit of humans. However, this should be done in a way and at a rate that does not lead to the long-term decline of biological diversity, thus promoting the sustainable use of biodiversity (SCBD, 2000).

Sustainability and sustainable use

The most commonly used definition of the term "sustainable development" is found in the 1987 report, Our Common Future, of the World Commission on Environment and Development (WCED, 1987). In this report sustainable development is defined as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs." Achieving this in practice requires that economic growth, social progress, and environmental quality improvement go together. These three pillars cannot be developed in isolation since they are strongly interdependent. Economic growth can provide the additional financial resources for improving the quality of the environment and reinforcing social cohesion. Social policy underpins economic performance and helps citizens to be responsible. Environmental policy contributes to preserving the natural resource base of the economy and to enhance the quality of life (CEC, 2003). Thus, the sustainable use of natural resources constitutes an effective tool to achieve sustainable development; achieving sustainable patterns of resource use is a key part of achieving sustainable development (SERI et al., 2009). However, it is not just the natural resources that must be sustainably managed. All components of ecosystems should be considered. Estuaries provide natural resources (e.g., fish and shellfish) but also have recreational and aesthetic value such as fishing, bird-watching, and boating. Additionally, estuaries are often the cultural centers of coastal communities, serving as focal points for local commerce, recreation, celebrations, and traditions (Figure 1).

In addition to the resources, sustainable use should also consider the ecosystems services (e.g., detoxification and decomposition of wastes, stabilization and moderation of the climate) (SCBD, 2000), which represent the benefits human populations derive, directly or indirectly, from ecosystem functions (Costanza et al., 1997).

Conditions for sustainable use

The challenge of sustainable use is the following: reversing the degradation of ecosystems while meeting the increasing demands for their resources and services. In 2004, the Secretariat of the Convention on Biological Diversity identified the fundamental conditions that should be taken into account in structuring a sustainable-use approach (SCBD 2004). These include the following:

1. Resources should be used in a manner in which ecological processes, species, and genetic variability remain above thresholds needed for long-term viability, and thus all resource managers and users have the responsibility to ensure that use does not exceed these capacities. It is crucial that the ecosystem is maintained or, in some cases, recovered, to ensure that those ecosystems are capable to sustain the ecological services on which both biodiversity and people depend.
2. Ecosystems, ecological processes within them, species variability, and genetic variation change over time whether or not they are used; therefore, governments, resource managers, and users should take into account the need to accommodate change, including stochastic events that may adversely affect biodiversity and influence the sustainability of a use.
3. Under circumstances where the risk of converting natural landscapes to other purposes is high, encouraging sustainable use can provide incentives to maintain habitats and ecosystems and the species within them.
4. Biodiversity provides many direct benefits and ecosystem services necessary for life. Increasingly, many marine species are of value to pharmaceuticals for disease prevention and cure. Finally, local communities and their cultures often depend directly on the uses of natural resources for their livelihoods. In all of these instances, governments should have adequate policies and capacities in place to ensure that such uses are sustainable.
5. The supply of biological products and ecological services available for use is limited by intrinsic biological characteristics of both species and ecosystems, including productivity, resilience, and stability.

Sustainable Use, Figure 1 Religious celebration in honor of *Nossa Senhora do Rosário de Tróia*, the patroness saint of fishermen of Setubal (Sado estuary, Portugal).

Biological systems, which are dependent on cycling of finite resources, have limitations with respect to the goods they can provide and services they can render. Although certain limits can be extended to some degree through technological breakthroughs, there are still limits and constraints imposed by the availability and accessibility of endogenous and exogenous resources.

6. To ameliorate any potential negative long-term effects of resource uses, it is incumbent on all resource users to apply the precautionary principle in their management decisions and to opt for sustainable-use management strategies and policies favoring uses that provide increased sustainable benefits.

Monitoring sustainability

Monitoring is the continuous or periodic process of collecting and analyzing data to measure the performance of a program, project, or activity. As an integral and continuing part of project/program management, monitoring provides managers and stakeholders with regular feedback on implementation and progress toward the attainment of environmental objectives (UNESCO, 2006). Monitoring enables management to take appropriate corrective action to achieve desired results. Effective monitoring requires baseline data, as well as indicators of performance and related measurements, regular reporting, and a feedback mechanism for management decision-making (UNESCO, 2006).

Effective monitoring and evaluation are widely recognized as an indispensable tool in assuring that the management objectives established are being achieved. If done well, a monitoring and evaluation plan and associated indicators serve both as a corrective function during the project cycle, enabling timely adjustments, and as a guide to structuring future projects more effectively.

Sustainable-use monitoring must involve the consideration of governance, ecological (including environmental) and socioeconomic dimensions, as well as the interaction between them (UNESCO, 2006). The indicators oriented to measure sustainable development are designated as sustainable development indicators. The recent report titled *Framework and Suggested Indicators to Measure Sustainable Development* (UNECE et al., 2013) presents an approach which aims to facilitate users' choices through large numbers of sustainable development indicators available in literature (e.g., EC, 2007; UN, 2007). Although this publication is primarily aimed at statisticians, it may also be relevant for policymakers, as policy targets for sustainable development are increasingly being formulated at national and international levels.

Estuaries

An ecosystem with unequaled value

Estuaries are highly productive ecosystems which provide a suite of resources and services (e.g., Nixon et al., 1986; Wilson and Farber, 2009; Barbier et al., 2011). Thus, estuaries are an irreplaceable natural ecosystem that must be managed carefully for the mutual benefit of all who enjoy and depend on them. Thousands of species of birds, mammals, and other wildlife depend on estuarine habitats as places to live, feed, and reproduce. And many marine organisms, including most commercially important species of fish, depend on estuaries at some point during their development. Estuaries are the year-round home for many species (e.g., oysters), while other species move in and out of estuaries on a seasonal basis for reproduction and growth (e.g., salmon and shrimp) (Wilson and Farber, 2009). Because they are biologically productive, estuaries provide ideal areas for migratory birds to rest and refuel during their long journeys. Additionally, numerous fish and invertebrate species rely on the sheltered waters of estuaries as nursery habitats (Vasconcelos et al., 2011). Estuaries have important commercial value and their resources provide economic benefits for fisheries, tourism, and cultural activities.

The protected coastal waters of estuaries also support important public infrastructure, serving as harbors and ports vital for shipping and transportation. Estuaries also perform other valuable services. They are inherently important to environmental and human health. Estuarine capacity to filter pollutants not only serves to provide a healthy environment for marine creatures to thrive, but it contributes to cleaner coastal waters for beach-going populations (Kildow, 2009). Wetland plants and soils also act as natural buffers reducing impacts by moderating the effects of stormwater runoff including stabilizing soil to prevent erosion; filtering suspended solids, nutrients, and harmful or toxic substances; and moderating water-level fluctuations (Castelle et al., 1992).

Use of estuaries: opportunities and threats

Estuaries are areas with major economic potential because of their strategic location close to seas and inland waterways. Estuaries also provide some of the world's most fertile areas for food production. That is why navigation and port development, as well as agriculture and fisheries, have always been the engines of economic development of estuaries. Attracted by these resources, large numbers of people live in the vicinity of estuaries leading to the growth of coastal cities and mega cities (Sekovski et al., 2012). Of the 32 largest cities in the world, 22 are located on estuaries (Ross, 1995). Five of the ten largest metropolitan areas in the United States are centered along major estuaries (NOAA, 1998 in Rice et al., 2005). For example, New York is located at the mouth of the Hudson River estuary; San Francisco is located on San Francisco Bay which is an estuary for the Sacramento and San Joaquin rivers; and New Orleans is on the estuary of the Mississippi River. Unfortunately, this increasing concentration of people disturbs the natural balance of estuarine ecosystems due to environmental impacts caused by development (Sekovski et al., 2012), which threatens their integrity, and imposes increased pressures on vital natural resources that endanger their susceptibility. For example, along the Hudson River, New York, human presence and activities have profoundly changed the estuary as a natural ecosystem (Figure 2).

Trends

Kennish (2002) describes the trends for estuaries until 2025. He suggests that estuaries will be most significantly impacted by habitat loss and alteration associated with a burgeoning coastal population, which is expected to approach 6 billion people. Habitat destruction has far-reaching ecological consequences, modifying the structure, function, and controls of estuarine ecosystems and contributing to the decline of biodiversity. Other anticipated high-priority problems are excessive nutrient and sewage inputs to estuaries, principally from land-based sources. These inputs will lead to the greater incidence of eutrophication as well as hypoxia and anoxia. During the next 25 years, overfishing is expected to become a more pervasive and significant anthropogenic factor, also capable of mediating global-scale change to estuaries. Chemical contaminants, notably synthetic organic compounds, will remain a serious problem, especially in heavily industrialized areas. Freshwater diversions appear to be an emerging global problem as the expanding coastal population places greater demands on limited freshwater supplies for agricultural, domestic, and industrial needs. Altered freshwater flows could significantly affect nutrient loads, biotic community structure, and the trophodynamics of estuarine systems. Ecological impacts that will be less threatening, but still damaging, are those caused by introduced species, sea level rise, coastal subsidence, and debris/litter. Although all of these disturbances can alter habitats and contribute to shifts in the composition of estuarine biotic communities, the overall effect will be partial changes to these ecosystem components.

Stevens (2010) also describes trends for estuaries but focuses on climate change effects (Table 1). With rising sea levels, estuaries will also be affected, causing changes in these waterbodies as manifested by the loss of intertidal area, erosion of shorelines, and increased risk of flooding of low lying areas (Rossington, 2008 *in* Stevens, 2010).

Given the above, management strategies must be developed to mitigate future impacts on estuaries.

The future of estuaries: managing, restoring and monitoring

It is a fact that increasing human activities in the coastal zone leave a significant human environmental footprint, leading to multiple stresses on estuaries and causing declines in water quality and overall ecosystem health. Included here are the effects of eutrophication, wastewater inputs, chemical contaminants, freshwater diversions, draining and ditching of wetlands, hardened shorelines, sediment/turbidity influx, inlet stabilization, introduced species, and fisheries overexploitation (Kennish, 2012).

One course of remedial action is habitat restoration. Restoring habitats involves reestablishing natural ecosystem processes by removing invasive species, reducing pollution levels, and reintroducing indigenous flora and fauna. The goal is to rebuild the estuary as an ecosystem that functions as it once did prior to impacts. Fortunately, estuarine ecosystems can often be restored, because of their adaptive and resilient capacity (Most et al., 2009). Simenstad and Bottom (2002) recommend the following principles that, although being developed in the context of estuarine habitat restoration for salmon recovery for

Sustainable Use, Figure 2 Hudson River, New York: Human presence and activities have profoundly changed the estuary as natural ecosystem.

the Columbia River system, can be generally applied to a wide range of restoration activities.

Protect first, restore second
Protection of existing habitat is critical to the success of estuarine restoration. To restore habitat in the absence of any overlying conservation program is counterproductive. All restoration sites should be explicitly incorporated into a broad conservation framework that will ensure their long-term protection. Estuaries and estuarine habitats can be restored only through a long-term stewardship approach with the necessary constituencies, policies, and funding to support it.

Do no harm
To ensure no net loss of habitat functions and to protect unimpeded natural processes, restoration actions should achieve proposed benefits without degrading other ecological functions of natural habitats or broader ecosystems.

Use natural processes to restore and maintain structure
Restorative measures should reestablish the dynamics of estuarine hydrology, sedimentology, geomorphology, and other habitat-forming processes that naturally create and maintain habitat, rather than simply implanting habitat structures at inappropriate or unsustainable locations. To the extent possible, engineered structures should be avoided in restoration designs.

Restore rather than enhance or create
Past experience demonstrates that, compared to restoration, enhancement (where designed to increase one or more specific functions of a degraded habitat) or creation of estuarine habitat is problematic and rarely leads to self-sustaining ecosystems.

Develop a comprehensive restoration plan using landscape ecology concepts to reestablish ecosystem connectivity and complexity
Unplanned, opportunistic approaches to restoration will not suffice. Strategic planning is necessary across hierarchical scales, from watershed to estuarine habitats, in order to set a broad vision, articulate clear goals, and place local restoration activities in an ecologically sound, ecosystem context. Both ecological and socioeconomic aspects of the estuarine landscape must be considered when selecting, designing, and locating restoration sites; restoration and conservation can only be effective if implemented within the human context. Restoration plans should be designed to restore ecosystem complexity and diversity. Public access to restoration sites should be

Sustainable Use, Table 1 Expect changes in estuaries from sea level rise (Stevens, 2010)

Changes within the environment	Changes within an estuary
Higher air temperatures	Increased evaporation and lower soil moisture affecting runoff to estuaries Increased fire risk for surrounding vegetation Increased stratification of coastal lakes
Decreased rainfall or changes to rainfall patterns	Decreased runoff and its impact on environmental flows Average rainfall might stay the same but how and when it falls could change, i.e., rain falling in very large storms less often Increased fire risk for surrounding vegetation
Sea level rise	Saline intrusion with dieback of freshwater wetlands Larger or more frequent storm surges impact barrier ridges and/or salinity of estuaries (wave-dominated estuaries) Inundation and shoreline recession including: ~ Erosion and landward recession of soft sandy shorelines, particularly where these are backed by low-lying plains of soft unconsolidated sediments ~ Modification of soft low-lying muddy estuarine and deltaic shores ~ Acceleration of existing progressive erosion of soft clayey-gravelly shorelines ~ Increased slumping of steep landslip-prone shorelines
Higher sea surface temperatures	Changes to nutrient cycling Changes to primary productivity Changes to water temperature of coastal waters
Ocean acidification	Changes to pH and pCO_2
Ocean circulation wave patterns	Changes to nutrient cycling Changes to sediment dynamics and form of estuaries
Vector-borne diseases	Change in the occurrence and distribution of vectors which utilize coastal waterways in their life cycles

encouraged wherever appropriate and incorporated into restoration plans. However, they should be designed to minimize impacts on the ecological functioning of the site.

Use history as a guide, but recognize irreversible change

Historic templates often provide the framework for restoration goals, as well as a perspective on how ecosystems have been incrementally degraded. Tidal, fluvial, geomorphic, and other naturally dynamic processes occur in a landscape context. Understanding the historic landscape structure is essential to comprehending how restoration can be implemented strategically in the modern landscape to promote the natural formation and maintenance of important habitats. Reconstructing the historical river, tidal floodplain, and estuarine structure does not necessarily guarantee restoration success but will decrease uncertainty.

Establish performance criteria based on specific objectives and monitor performance both individually and comprehensively

Monitoring and adaptive management are essential components of restoration and habitat management. Objectives for restoration projects should be clearly stated, site specific, measurable, and long-term, in many cases greater than 20 years. Performance criteria should derive directly from these goals and should include both functional and structural elements and be linked to suitable, local reference ("target") habitats. Scientific monitoring based on the established performance criteria is essential to improve restoration techniques and to achieve estuarine restoration goals. All restoration designs should be monitored and, based on the concept of adaptive management, altered if necessary to achieve desired end points and to insure that local projects are self-sustaining.

Use the best interdisciplinary science and technical knowledge and use science review processes

All available scientific and technical expertise should be brought to bear on the complex problems of estuarine habitat restoration. Restoration should be planned, designed, implemented, and monitored by an interdisciplinary, not just multidisciplinary, group of experts. Physical (e.g., hydrology, geomorphology, geophysics, sedimentology), chemical (e.g., sediment geochemistry), mathematical (e.g., biostatistics), and engineering sciences should be represented in addition to the essential biological disciplines (e.g., estuarine and fish ecology, landscape ecology, botany). An independent, peer-review panel should be established to evaluate scientific assumptions and performance throughout the restoration process and to ensure restoration goals are achieved.

These recommendations stress the importance of monitoring. Monitoring is essential for assessing whether the action led to the hypothesized result and for providing managers and researchers with increasing knowledge about the feasibility and approaches to rehabilitation (Rice et al., 2005). According to Thom and Wellman

Sustainable Use, Table 2 Examples of estuarine rehabilitation activities by ecosystem type (Rice et al. 2005)

Ecosystem	Actions
Tidal marsh	Dike or levy breach or removal, excavation, substrate addition, transplantation, fertilization, hydrologic control (e.g., tide gates), grazer control, competitor control, large woody debris placement, wastewater and sediment discharge control, chemical contaminant removal or containment
Sea grass	Transplantation, fertilization, excavation, substrate addition, wastewater and sediment discharge control, chemical contaminant removal or containment
Kelp	Transplantation, substrate addition, grazer control, competitor control, wastewater and sediment discharge control
Mudflat	Dike or levy breach or removal, excavation, substrate addition, chemical contaminant removal or containment
Sand/gravel beach	Substrate addition, excavation

(1996), monitoring should be considered as a fundamental part of a restoration project aiming to: (1) assess the performance of the restoration project relative to the project goals, (2) provide information that can be used to improve the performance of the project, and (3) provide information to interested parties.

Rice et al. (2005) showed that the most common technique in estuarine rehabilitation is the return or the introduction of tidal inundation. Table 2 lists some examples of estuarine rehabilitation activities by ecosystem type.

Rice et al. (2005) also present examples of physical, biological, and chemical controlling factors and structural and functional attributes for use in estuarine rehabilitation monitoring (Table 3).

The way forward

Population growth, economic development, and climate change are factors that can significantly impact estuaries. Efficient management of natural resources and services is a key to maintaining the integrity of estuaries. To achieve this, innovations are required – social, institutional, and technological innovations (Most et al., 2009). Most et al. (2009) have examined approaches for achieving sustainable development of estuaries, as recounted in the four passages below.

Estuarine vision: a shared view on sustainable development

A shared vision on estuarine sustainable development should deal with all drivers of change in an estuary (population growth, economic development, and climate change) as well as with the relevant societal trends (decentralization, privatization, participation, growing environmental concerns, and growing risk aversion). Such a vision should be developed in close cooperation with all parties that have a stake or a say in the development of the estuaries.

Estuarine technology: innovations in science and technology

Sustainable development of estuaries requires innovations in the knowledge of natural systems' behavior as well as in the approach to planning and design. An important source of innovation is the development in information and communication technology. Advances in sensor and simulation technologies may promote the development of more accurate warning and forecasting systems. These technologies also support the development of local- and global-scale monitoring and diagnostic systems. Integration of knowledge from various disciplines may open new applications as well.

Estuarine governance: social and institutional innovations

For development of estuaries to be more sustainable, it is important to obtain societal acceptance and support for this development. Good governance should foster shared visions on sustainable development of estuaries. Moreover, conditions should be created for the actual implementation of such visions through development projects. Governance should also provide adequate arrangements for maintenance of infrastructure to prevent early deterioration of the infrastructure.

Estuarine dialogue and forum: learning from others

Sustainable development of estuaries is an increasingly complex field which requires the contribution and cooperation of many parties. Although there is no general strategy on how to best deal with many estuarine issues, it is important to learn from experiences elsewhere. To this end, the exchange of knowledge and experiences should be encouraged.

Best practices in the sustainable use of estuaries

Estuaries have characteristics in common, but there is also much diversity in physical conditions, governance structure, and cultural background. Hence, there is no general approach on how to deal with estuarine issues. Nevertheless, some broad perspectives may be generated to deal with these issues. Most et al. (2009) identified the emerging "best practices" for deltas (that can be generalized for estuaries) which should comprise a balanced mix of measures from the different response themes and reflect the integrated nature and regional scale of estuarine development.

Relieving the pressure on available space

Spatial planning regulation may relieve some of the pressure by redirecting urban development and economic activities to less "crowded" and/or low-risk areas. In cases where spatial planning offers little solace, land

Sustainable Use, Table 3 Examples of physical, chemical, and biological variables for controlling factors and structural and functional attributes that could be considered as potential metrics in estuarine rehabilitation monitoring (Rice et al. 2005)

Category	Controlling factors	Structural attributes	Functional attributes
Physical			
Hydrology	Geomorphology, freshwater inflow, tidal regime	Tidal range, tidal prism, hydroperiod, residence time	Fish presence/absence (access to habitat)
Geomorphology/ topography	Geology, tidal regime, sedimentation	Elevation, connectivity, channel complexity	Fish presence/absence (access to habitat)
Water characteristics	Freshwater inflow, tidal regime, nutrient concentrations, biochemical oxygen demand, residence time	Temperature, salinity, dissolved oxygen (DO), current stratification	Fish prey production (capacity of habitat)
Soil/sediment	Geology, tidal regime, sediment supply	Grain size, organic carbon content, nutrient concentrations, salinity, redox potential	Sedimentation, organic carbon accumulation, nutrient accumulation
Chemical			
Nutrients and organic matter	Freshwater runoff, point and sources, marine upwelling, sedimentation	Nutrient concentrations, organic carbon content	Primary production, invertebrate community structure and production
Contaminants	Point and nonpoint sources organic carbon, hydrology	Chemical concentrations in sediment, water, and biota	Altered organism growth, reduced immune function
Biological			
Emergent vegetation	Elevation, tidal regime, salinity, soil composition, pore water salinity, competition, grazers	Area, percent coverage, shoot density, biomass, height, species richness, relative abundance	Primary production, faunal utilization
Submergent vegetation	Elevation, substrate, light, temperature, salinity, nutrients, flow	Area, percent coverage, shoot density, biomass	Primary production, faunal utilization
Benthic invertebrates	Substrate, elevation, temperature, salinity, DO, chemical contaminants	Abundance, species richness, relative abundance, dominance	Biomass, presence in predator diet
Fishes	Temperature, salinity, DO, access, flow, food availability, predation, competition, harvest	Abundance, species richness, relative abundance, dominance	Growth, fecundity, residence time, movement patterns, survival, population structure, population growth
Birds	Access, food availability, nesting site availability, predation, competition	Abundance, species richness, dominance	Growth, fecundity, residence time, survival, behavior, population structure, population growth
Phytoplankton	Light, temperature, salinity, nutrients, stratification	Abundance, species richness, dominance	Primary production
Zooplankton	Temperature, salinity, DO, flow, phytoplankton	Abundance, species richness, dominance	Density, biomass, presence in predator diets

reclamation has proven to be an effective way to relieve some of the pressure on space. Land reclamation also offers good opportunities for implementation of the "building with nature" concept, meanwhile easily applying new safety considerations. Multifunctional use of areas, e.g., giving a water storage function to nature areas, may further assist in relieving the pressure on space.

Improving resilience of estuaries

Vulnerability of societies to future climate change (such as flood risks, droughts, and salinity intrusion) should be reduced, preferably by making societies more resilient. Resilience can be improved by preparedness, coping strategies, and adaptation to changing conditions. This requires a combination of willingness to change, appropriate technology, and community participation. Increasing the robustness of infrastructure is another promising way to respond to the increase of vulnerability of estuarine areas as well as the growing aversion of risk.

Secure fresh water supplies

Many estuaries in the world currently face water shortages which may be worsened by climate change and pollution. Adaptation to land and water use will be an important way to respond to these shortages. This may include more efficient water use and/or changes in cropping pattern and fertilization in agriculture. Pollution reduction programs and establishment of flow requirements for estuaries are needed. Their implementation may benefit by involvement of river basin agencies.

Upgrade aging infrastructure

Many estuaries have irrigation and drainage systems as well as flood protection works, roads, water supply, and treatment facilities which require upgrading. Public and private partnerships can provide solutions in those cases where farmers, industries, and communities directly benefit from these infrastructure investments. However, for protection schemes against floods and storm surges, other options may be more appropriate, such as introducing financing mechanisms. Restoring infrastructure also offers opportunities for multifunctional use of the infrastructure.

Coastal erosion management

Many estuaries are experiencing coastal erosion problems due to sediment shortages. Solutions should preferably include a restoration of the sediment balance. If this is not feasible, sand nourishments are preferred over hard engineering structures. Also, other "building with nature" options should be looked into, e.g., mangrove restoration. This is primarily a task for coastal management agencies, which should work closely together with local stakeholders and the private sector.

Biodiversity protection and restoration of ecosystems

Estuarine ecosystems are under significant pressure worldwide due to human activities. Effective action must be undertaken to protect against habitat destruction, external disturbance, and pollutant inputs. This requires adhering to national and international controls such as the Habitats Directive, Ramsar Convention, and Convention on Biological Diversity. Biodiversity protection should be implemented at the local level through cooperation and involvement of all stakeholders. Efforts should be made to restore estuarine ecosystems. An integral approach and early involvement of stakeholders contribute to the success of restoration efforts. The integrity of (altered) estuarine ecosystems may be enhanced through reconnection with rivers and seas.

Summary

Ecosystem resources and services are vital for the economic growth and sustainability of coastal communities. However, estuaries are facing greater pressures worldwide due to rapid population growth, economic development, and climate change. This places greater importance on effective management of estuaries. To promote sustainable development of estuaries, a clear vision and direction must be developed on how to best respond to the various drivers of change as well as to address the needs and trends of society. Thus, it is necessary to reverse unsustainable trends by shifting to more sustainable-use patterns, thereby enhancing vital environmental functions, economic growth, and social progress.

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Cross-references

Adaptive Management
Coastal Wetlands
Estuarine Habitat Restoration
Estuary Conservation

SYMBIOSIS

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Definition

Symbiosis is defined as a lasting, intimate association between members of different species of organisms (de Bary, 1879). There are mutualistic, parasitic, and antagonistic symbioses. Etymology: From Greek σύν – together and Greek βίος – life.

Description

Interspecific cohabitation occurs in a wide range of habitats and across all known forms of life. The character of species interactions may change over time or with conditions (Aanen and Hoekstra, 2007; Roossick, 2011). Beyond beneficial or detrimental effects among symbiotic partners, symbioses can alter ecosystems and affect biogeochemical cycles (Dziallas et al., 2012).

The close physical association of symbionts can facilitate horizontal gene transfer, which allows for genetic responses (adaptation) to environmental change. In addition, the inactivation or termination of existing symbioses in favor of activating or establishing different ones may add flexibility to the ecophysiological properties of the symbiotic community (Baker, 2003; Dziallas et al., 2012).

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Cross-references

Mutualism

T

TECTONIC EUSTASY

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Synonyms

Tectono-eustasy

Definition

Tectonic eustasy (often written, *tectono-eustasy*) denotes the sea-level changes generated by changes in ocean basin volume (e.g., Mörner, 1986; Mörner, 1987; Rona, 1995).

Description

Changes in ocean basin volume are generated by different geodynamic processes such as sea-floor subsidence, ridge growth, plate motions, and crustal movements. It is a slow process leading to maximum sea-level changes of ~0.06 mm/year (Mörner, 1996). Global isostatic adjustment (GIA) in response to the glacial loading/de-loading in association with ice ages (e.g., Peltier, 1982, 1998) may be considered a special element of tectonic eustasy.

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Cross-references

Shoreline Changes
Submerged Coasts
Submergent Shoreline

THERMAL BIOLOGY

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Synonyms

Biophysical ecology; Organism temperature; Thermal physiology

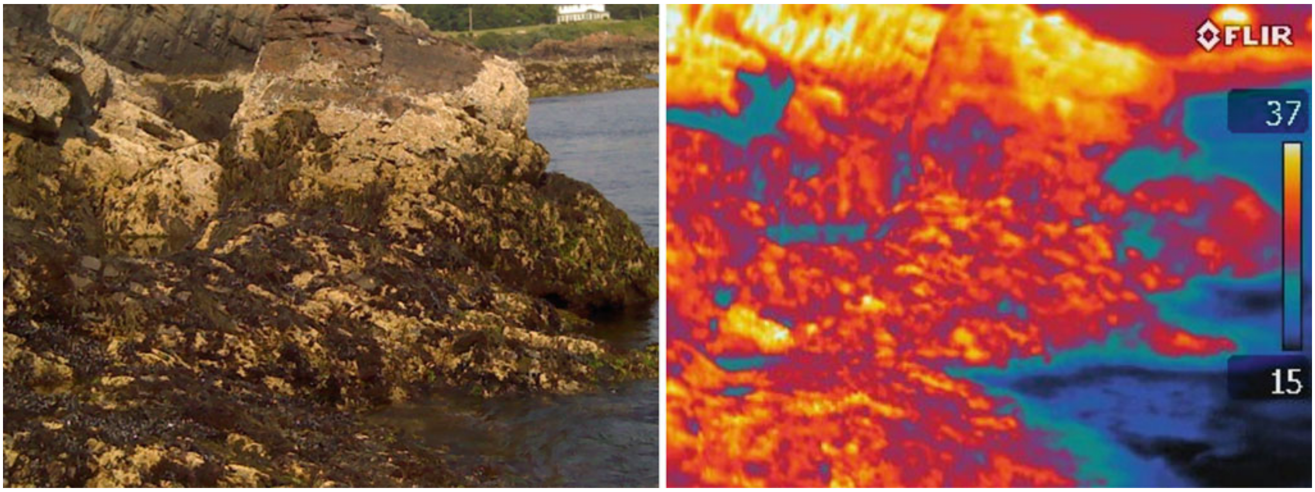
Definition

Thermal biology is the study of physiological and ecological consequences of body temperature and of the biophysical, morphological, and behavioral determinants of organism temperature.

Introduction

The temperature of an organism's body affects almost all physiological processes, which in turn can have significant consequences for organisms, and interactions such as competition and predation. While some animals (and a few plants) make significant amounts of metabolic heat

M.J. Kennish (ed.), *Encyclopedia of Estuaries*, DOI 10.1007/978-94-017-8801-4,



Thermal Biology, Figure 1 Photographic image of the intertidal community (*left*) demonstrates species vertical zonation and substrate heterogeneity, with concurrent infrared (IR) thermal image (*right*) of the same site during low tide. The IR image demonstrates the large variability in temperatures (15–37 °C) observed in this small section of shore, which may in part drive small-scale distribution and zonation patterns. Photograph courtesy of N. Colvard and B. Helmuth, FLIR E60 Infrared Camera.

(i.e., are endothermic), the body temperatures of estuarine invertebrates and plants are driven by the exchange of heat with the ambient environment – they are ectothermic – and can vary substantially over the course of a day, season, and year.

Heat exchange and body temperature

Intertidal organisms in estuaries are subject to diel tidal fluctuations, and so their body temperatures at low tide aerial exposure can significantly differ than that during submergence (Jost and Helmuth, 2007). The dominant driver of heat input in these systems is solar radiation, so that the temperature of an animal's or a plant's body is generally much warmer than the temperature of the surrounding air during the day; at night, as heat is lost through infrared radiation, organisms can be several degrees cooler than the air. The amount of shading can have a large influence as can the amount of wind (convection). Thus, for example, snails at the top of a marsh canopy can have temperatures different from animals near the substrate (Iacarella and Helmuth, 2011). Solar heating can also increase the temperature of shallow waters so that continually submerged estuarine organisms can in some cases experience large fluctuations in temperature.

The behavioral and morphological characteristics of an organism dictate the rate of heat transfer between the surrounding environment and the organism, and the size, shape, mass, and color of an organism can affect body temperature. Thus, two organisms exposed to identical environmental conditions can display very different body temperatures from one another, even when neither is making any appreciable metabolic heat (Figure 1). The behavior of an organism can also affect rates of heating and cooling. Organisms such as algae and sea stars with wet surfaces lose considerable amounts of heat via the evaporation of water, although this water loss potentially comes at the cost of desiccation (Iacarella and Helmuth, 2011). As above, the movement of organisms into shaded crevices, on the shaded sides of rocks, or depth in the sediment can lead to temperature differences over the scale of centimeters that exceed those observed over thousands of kilometers of latitude (Denny et al., 2011).

Direct and indirect effects of body temperature on estuarine organisms

Organism temperature has numerous effects on physiological processes in estuarine organisms (Somero, 2011), including functioning of critical enzymes and other cellular functions, changes in metabolic rate, and increases in metabolic oxygen demand (Pörtner, 2010). For an excellent overview of physiological impacts of temperature on marine organisms, see Somero (2011). The thermal biology of estuarine organisms also indirectly affects species through interactions such as competition and predation (Russell et al., 2012). Wethey (1984) showed that the competitive ability of barnacles is affected by their temperatures and that the zonation of two barnacle species could be explained by their differential physiological sensitivity to body temperature. Sanford (2002) demonstrated that rates of predation by the keystone sea star *Pisaster* were elevated at increased water temperature, and conversely, Pincebourde et al. (2008) reported decreases in predation rate by up to 40 % during elevated temperatures at low tide in air.

Monitoring temperature

In water, the temperature of a marine organism approximates that of the surrounding water, although in some cases of low water flow and high solar radiation, temperatures of organisms such as corals can be several degrees warmer than the water around them (Jimenez et al., 2008). Gradients in water temperature in many coastal systems can be very steep, so that the temperature even a few meters below the surface can be considerably cooler than that recorded on the surface, for example, by satellite (Castillo and Lima, 2010). Measuring the temperatures of organisms in situ and at scales appropriate to the organism in question is therefore vital for understanding their thermal biology.

During aerial exposure at low tide, animal and plant temperatures are often quite different from those of the surrounding air and substrata (Lathlean et al., 2011). Frustratingly, the same factors that lead to observed differences in organism temperature also affect the temperature recorded by instruments, such as now commonly used data loggers, so that the size, color, and mass of the instrument (or worse yet, the casing in which it is housed) all affect the temperature that it records. Fitzhenry et al. (2004) reported average differences of 14 °C between the body temperature of intertidal mussels and the adjacent temperature loggers. To combat this issue, the use of biomimetic temperature loggers has been adopted (Lima et al., 2011). Biomimetics are "thermally matched" to organisms and can in many cases record temperatures close to that of study organisms (Lima et al., 2011). Notably, a central message is that a single instrument almost never records measurements that are applicable to all organisms, since each can display a different temperature.

The use of infrared thermography (Figure 1) has also been used to measure patterns of temperature in the field (Lathlean et al., 2012), although this approach carries with it some drawbacks as well, as the camera needs to be calibrated to the surface properties (emissivity) of each species and the instrument cannot be deployed continuously for long time periods.

Climate change and thermal biology

The relevance of thermal biology in coastal regions has received renewed attention in the face of climate change. Changes in the distribution and abundance of many coastal species have been reported (Southward et al., 2005). These studies have emphasized that if we are to predict the likely impacts of ongoing and projected climate change on coastal species, a nested approach that considers the physiological, ecological, and biogeographic mechanisms by which the environment affects the thermal biology of organisms in what are likely to be unprecedented conditions should be used (Howard et al., 2013).

Summary

The study of thermal biology in estuarine species is very important since these organisms are subject to daily fluctuations in environmental conditions from ebb and flood tides. Estuarine organisms can serve as model systems when evaluating the impact of climate change and associated environmental processes because of the unique interchange with terrestrial and aquatic environments. In some localities estuarine organisms are living at their thermal maximum, and with the ensuing threat of aerial and water temperature increases in the coming century, it will be paramount to understand how these changes will impact the primary producers and ecosystem engineers in estuarine biotic communities.

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Cross-references

Climate Change
Ecological Niche
Environmental Gradients
Tides

TIDAL ASYMMETRY

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Definition

Tidal asymmetry is defined by the discrepancies in the duration of the falling and rising tides of water elevation. This duration asymmetry may be manifested as an inequality in flood/ebb tidal current magnitudes.

Description

Tidal asymmetry can produce two types of estuarine systems: flood dominant or ebb dominant. The system referred to as flood dominant occurs when the duration of the falling tide exceeds that of the rising tide leading to a larger peak flood current (Headquarters, U.S. Army Corps of Engineers, 2002); when the duration of the falling tide is shorter than that of the rising tide, it leads to a stronger peak ebb current, and then the system is referred to as ebb dominant.

The sources behind the asymmetrical tides are the interactions of multiple tidal constituents (Boon and Byrne, 1981). Phase difference between constituents dictates the direction of asymmetry (i.e., flood or ebb dominance), while the ratio of constituent amplitudes reflects the degree of distortion (Friedrichs and Aubrey, 1988).

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Cross-references

Tidal Hydrodynamics
Tidal Ranges

TIDAL DATUM

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Definition

A tidal datum is a reference elevation of sea level, defined in terms of a certain phase of a *tide*. Tidal datums are primarily used to estimate water heights and/or depths and determine horizontal boundaries. Tidal datums are fundamental to the determination of spatial coordinates of latitude, longitude, and elevation relative to mean sea level. They are also essential as legal entities to establish the limits of state jurisdiction over maritime zones (NOAA, 2000).

Description

Tidal datums are local because they are estimated from water level observations made at a particular tide gauge site. Tidal ranges vary considerably spatially depending on local hydrodynamic and topographic characteristics (e.g., tidal range may be much larger at the head of an estuary compared to the mouth), and hence tidal datums vary accordingly. Therefore they should not be extended into areas with significantly different oceanographic characteristics without sufficient observations (or numerical modeling) to support any such extrapolation.

Tidal datums are usually estimated statistically from water level observations made over a tidal datum *epoch*, which is the interval of time recommended to calculate tidal datums. This interval is usually at least 19 years, to ensure the full 18.6-year lunar nodal cycle is included in the analysis. The tidal datum is the average water level over a tidal epoch of water height at a particular tidal phase.

A multitude of tidal datums are currently in use within different counties. The common tidal datums are:

- Highest astronomical tide (HAT): The highest water level predicted to occur under any combination of astronomical conditions.
- Mean high water springs (MHWS) and mean low water neaps (MHWN): The average of all high waters at the time of spring or neap tide, respectively. This is applicable only in regions with semidiurnal tides. In mixed and diurnal tidal regions, mean higher high water (MHHW) and mean lower high water (MLHW) are used, which are the mean of the higher or lower of the two daily high waters, respectively.
- Mean high water (MHW): The average of all high waters.
- Mean sea level (MSL): The arithmetic mean of hourly observations of water level.
- Mean low water springs (MLWS) and mean low water neaps (MLWN): The average of all low waters at the time of spring or neap tide, respectively. In mixed and diurnal tidal regions, mean higher low water (MHLW) and mean lower low water (MLLW) are used, which are the mean of the higher or lower of the two daily low waters, respectively.
- Mean low water (MLW): The average of all low waters.
- Lowest astronomical tide (LAT): The lowest water level which can be predicted to occur under any combination of astronomical conditions. LAT is often used to define chart datum (CD), which is the water level which depths displayed on a nautical chart are referenced to.

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Cross-references

Mean Sea Level
Tidal Ranges
Tides

TIDAL FLAT

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Synonyms

Mud flat; Sand flat

Definition

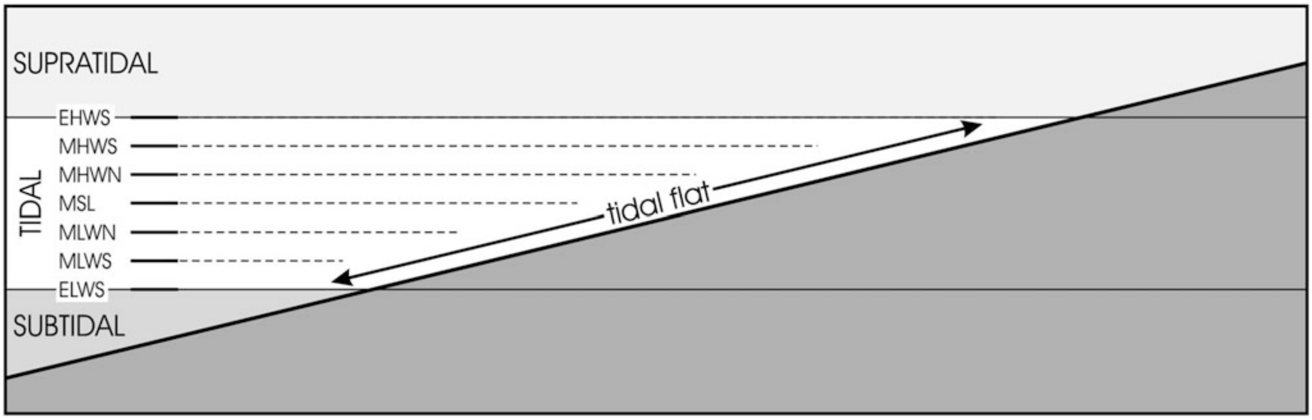
A tidal flat is a low-gradient surface, underlain by mud, muddy sand, or sand, which is exposed during low tide.

Definitional limits, zones, and characteristics of tidal flats

Before ecological aspects of tidal flats are described, some background information on the variety of tidal flats is provided from the perspective of their limits, zones, geomorphology, sedimentology, hydrology, and salinity, as these will determine, for a given climate, the types of habitats developed, the biota inhabiting them, and ecological processes that will operate therein.

The concept and definition of a tidal flat varies with author and between disciplines. Tidal flats, in the broadest sense, are tidally exposed low-gradient surfaces, underlain by a variety of substrates, and variably vegetated depending on climate and location of the surface relative to mean sea level (Semeniuk, 2005). They can be free of vegetation or inhabited by mangroves, saltmarsh, or seagrass. In the narrowest sense, for some authors, tidal flats are the vegetation-free tidally exposed low-gradient surfaces usually occurring between MSL and low tide though, even in this case, these types of flats may have a sparse ground cover of seagrass. Some authors may incorrectly exclude flats vegetated by mangroves (mangal) and saltmarsh, but the vegetated parts of tidally exposed low-gradient surfaces are still tidal flats. Although they have the same geomorphic expression as traditional tidal flats, extensively exposed rock pavements cut to low-gradient flats in the intertidal zone, and gravel-floored flats are generally not considered to be tidal flats. The consensus on tidal flats is that the tidally exposed surface is underlain by soft sediment (sand, muddy sand, or mud).

The moon and sun, the rotation of the earth, the regional bathymetry, and the shape of the coast are major factors in determining tidal hydrodynamics, the magnitude of tides, and the forcing of the ebb and flow of tides (Cartwright, 1999). Water levels also can be affected by barometric pressure and wind. The major effect of the moon and the sun is to generate spring tides (the larger tidal oscillations approximately coinciding with a new moon and full moon or sun-earth-moon syzygy) and neap tides (the smaller tidal oscillations approximately coinciding with the first quarter moon and third quarter moon). Tidal zones are identified as follows: equinoctial low water spring tide (=ELWS), a spring tide low water occurring twice yearly during the equinoctial period; mean low water spring tide (=MLWS), the mean position of low water of spring tides; mean low water neap tide (=MLWN), the mean position of low water of neap tides; MSL, the mean position of water level between low water and high water; mean high water neap tide (=MHWN), the mean position of high water during neap tides; mean high water spring tide (=MHWS), the mean position of high water during spring tides; and equinoctial high water spring tide (=EHWS), a spring tide high water occurring twice yearly during



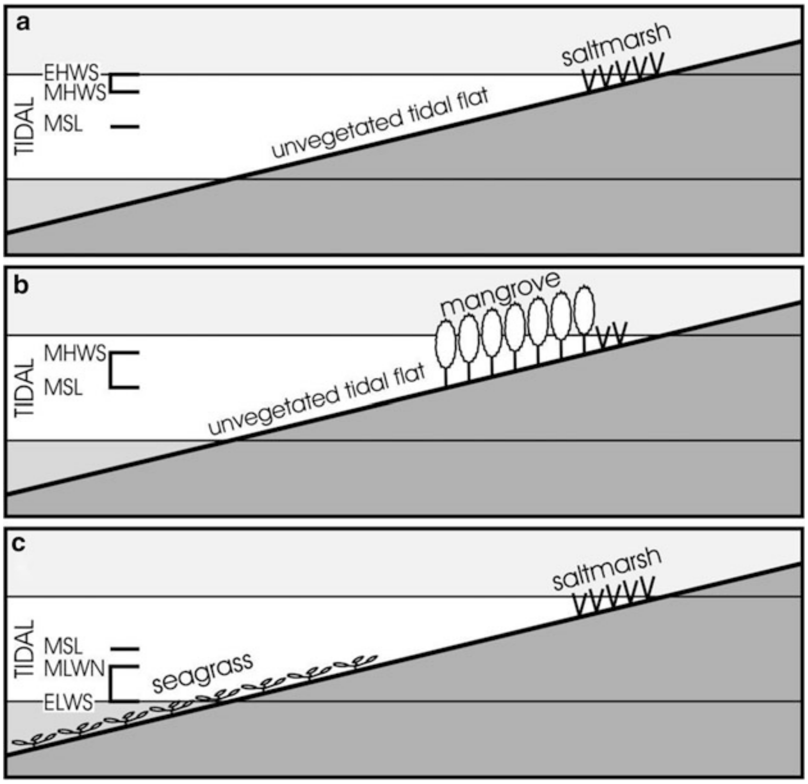
Tidal Flat, Figure 1 Tidal flat in profile showing the various tidal levels. Located between the supratidal and subtidal zones, the tidal flat has the following levels: the lowest tidal level at equinoctial low-water spring tide (ELWS), mean low-water spring tide (*MLWS*), mean low-water neap tide (*MLN*), mean sea level (*MSL*), mean high-water neap tide (*MHN*), mean high-water spring tide (*MHWS*), and the highest tide level at equinoctial high-water spring tide (*EHWS*).

the equinoctial period. These various levels are shown diagrammatically in Figure 1. In this context, the entire surface between ELWS and EHWS is the tidal zone, and any low-gradient surfaces between these levels are tidal flats. Above EHWS is the supratidal zone, and below ELWS is the subtidal zone. During and the 1960s 1970s, geological papers describing tidal flats in some regions incorrectly termed the spring-tide-flooded surface between MHWS and EHWS that dried out during the neap tides as the "supratidal" zone. It should be noted that the tidal zones delineated by tidal levels on a tidal flat may not necessarily encompass the entire tidal interval (i.e., the extent of a tidal flat may not equate to the full extent of the tidal zone) and the tidal flat may be confined, for instance, to the interval between ELWS and MSL, or EHWS and MHWN. Tidal intervals that are inhabited by mangroves, saltmarsh, or seagrass or that are vegetation-free are shown diagrammatically in Figure 2.

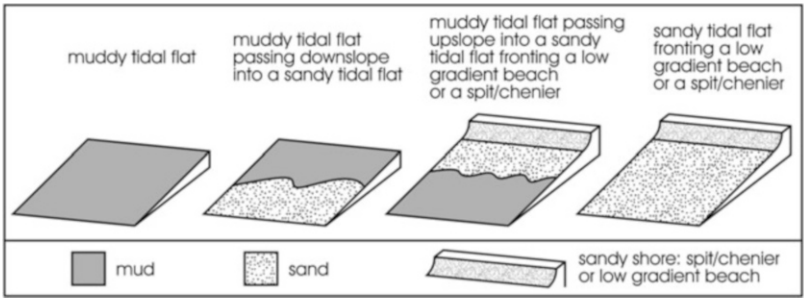
Tidal flats can directly adjoin a hinterland upland (and be bordered along their landward edge by a cliff, by a relatively more steeply sloping supratidal shore, or by a dune) or adjoin a spit, sand bar, chenier, or sandy beach. Rather than always being a simple uniformly sloping surface from high water to low water, tidal flats often have variation in gradients across the tidal interval (such as low hummocks of sand, meters in width, on a mud flat, or a low-relief chenier), as well as variation in substrate types. The variation in slope is related to tidal level, tide and wave energy gradients, and vegetation. Where there is a range of sediment grain sizes from mud to sand on the tidal flat, the dominant grain size, degree of exposure to low-amplitude wave action when inundated, and the tidal range, which determines current energy, will determine whether tidal flats are narrow or wide and whether they are wholly sandy, wholly muddy, grade from sand to mud from low tidal levels to high tidal levels, or the reverse from mud to sand from low to high tidal level (Figure 3). The frequency of inundation and the magnitude of tides determine the magnitude of tidal currents and, hence, substrate type. For example, microtidal areas generally have low-velocity tidal currents, while macrotidal areas generate large currents. Further, with wave action, mainly on sandy tidal flats, the tidal flat can be shaped into shore-parallel, low-relief linear shoals (or bars) and depressions, or low-relief shore-normal sand bars. As a result of tidal current erosion, the tidal flat can be traversed by dendritic to meandering shore-normal tidal creeks. These smaller-scale variations influence the distribution of biota across the tidal flat and the development of small-scale habitats. Vegetation such as saltmarsh and mangrove, in trapping sediment, can alter the slope of the tidal flat in that there is a relative buildup of sediment in the vegetated zone.

During low tide, many sandy tidal flats are covered by a very thin film of mud as there generally is a small proportion of mud in suspension in the waters flooding a tidal flat which will settle, giving the impression that the tidal flat is underlain by mud. While there are true mud flats, many tidal flats that have been termed "mud flats" actually are sandy tidal flats with a thin film of mud.

When tidal flats are exposed during low tide, water under, or within a flat is subject to evaporation and transpiration resulting in salinization, and a tidal flat salinity gradient is developed. This is systematic, progressive increase in salinity in groundwater (as pore water) and pellicular water (the water circumferential to sediment grains in a temporary vadose zone) across the tidal flat. The salinity gradient is described by Semeniuk (this volume) in "Tidal flat salinity gradients." The change in salinity across a tidal flat is due to the interacting effects of inundation, evaporation, transpiration, and sediment types. The salinity of groundwater and pellicular water varies spatially and temporally depending on (1) the position of a tidal flat in an estuary (and hence the salinity of the recharging source water); tidal flats



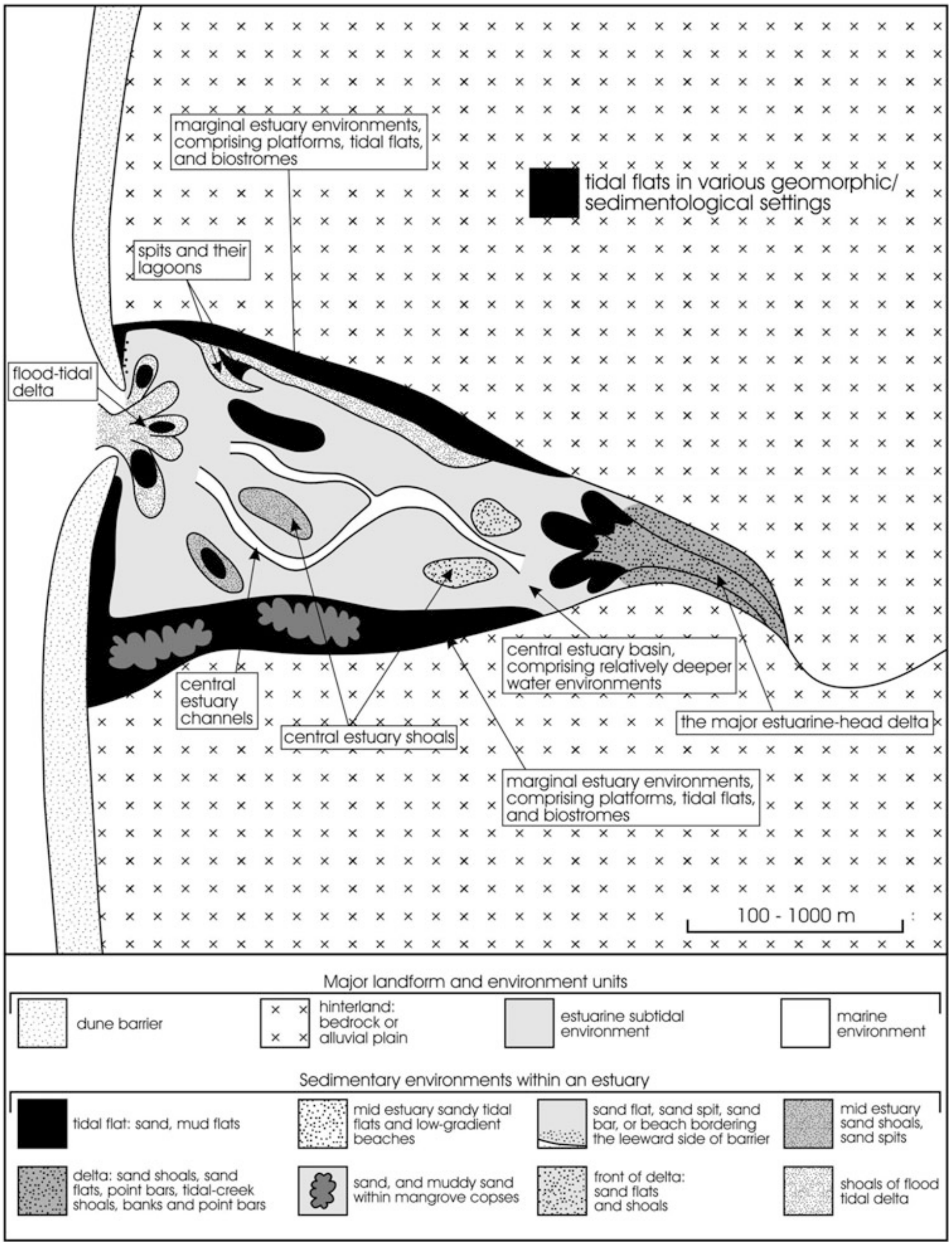
Tidal Flat, Figure 2 Idealized diagram of tidal flat surfaces showing a range of vegetation and plant life that may occupy specific tidal zones. The positions of the various tide levels for these profiles are shown in Figure 1. (**a**) Saltmarsh on the high tidal flats usually between MHWS and EHWS. (**b**) Mangroves on the high tidal flats usually between MSL and MHWS, bordered in this example by saltmarsh on the landward side. (**c**) Seagrass between MLWN and subtidal zone, and saltmarsh between MHWS and EHWS.



Tidal Flat, Figure 3 Idealized diagram showing a range of tidal flats underlain by mud, or mud grading downslope to sand flats, muddy tidal flat grading upslope to sand flats fronting a low-gradient beach or sandy spit/chenier, and sandy tidal flat fronting a low-gradient beach or sandy spit/chenier.

located in seaward parts of an estuary tend to be more saline in groundwater and pellicular water than those located in the riverine parts; (2) location on the tidal flat relative to MSL; (3) the stage of the tidal cycle in terms of spring tide versus neap tide, and rising tidal water versus falling tidal water; (4) the length of exposure of the tidal flat within the tidal cycle or between spring tides and neap tide phases; and (5) the climate, which determines the extent of evaporation, the influence of rainfall on the tidal flat, and the development of mangroves and saltmarsh. The salinity of the tidal flat groundwater tends to be relatively constant from tide to tide during a given season and can even be constant over the entire year. Transpiration by tidal flat vegetation increases baseline salinities of pellicular and groundwaters.

While there may be a general trend of increasing tidal flat salinity from low tidal levels to the high tide mark, this can be reversed by freshwater seepage along the margin of



Tidal Flat, Figure 4 Diagram showing the typical geomorphic and sedimentologic location of tidal flats within an idealized estuary.

the hinterland, from beaches, and from spits and cheniers (Semeniuk, 1983). Freshwater seepage can manifest as discharge along the upper tidal flat surface as surface flows, springs, or as subterranean flow along appropriate stratigraphic conduits or sheets (Semeniuk, 1983; Cresswell, 2000; Semeniuk et al., 2011).

Tidal flats in an estuarine environment

Tidal flats occur in different and varied geomorphic and sedimentologic settings within an estuary. The main settings are platforms bordering the margins of estuaries, tidal parts of flood-tidal deltas, tidal parts of intra-estuarine riverine deltas, the crests and flanks of middle-of-estuary shoals, and the protected lagoons leeward of spits. Figure 4 diagrammatically shows the locations of tidal flats within an idealized estuary.

The platforms bordering the margins of estuaries, since they can occur continuously or discontinuously along the whole length of an estuary, can traverse salinity fields from marine (at seaward parts of an estuary) to brackish or even freshwater (at riverine parts of an estuary). In terms of substrates, they can be sandy, muddy sand, or muddy, depending on sediment supply and local hydrodynamics. The tidal parts of flood-tidal deltas tend to be underlain by sand and have marine salinities. The tidal parts of intra-estuarine riverine deltas can be brackish to freshwater, varying between the seasons (and even marine water during the dry season) and, depending on the nature of the delta, can be underlain by sand, muddy sand, or mud (see *Estuarine Deltaic Wetlands* and *Deltas*). The crests and flanks of middle-of-estuary shoals reside in water that varies from marine at seaward parts of the estuary to brackish/freshwater in riverine parts of an estuary. Again, depending on sediment supply, the tidal flats can be underlain by sand, muddy sand, or mud. The tidal parts of protected lagoons, leeward of spits, tend to be in seaward parts of the estuary (though they can occur widely) and, as such, waters usually are marine salinity. Their substrates are sandy. However, if there is abundant mud in suspension in the estuarine waters, the low-energy protected leeward zone results in mud accumulation and the tidal flats are mud floored.

With all these tidal flats, except generally for the flood-tidal delta, if there is both sand and mud available, the sand and mud commonly are partitioned in different parts of the environment because of hydrodynamic processes, and as such, there can be depth-related mud tidal flats grading to sand tidal flats.

Biota of tidal flats within estuaries

There are various factors that influence the structure and composition of biota that inhabit a tidal flat. These include biogeography and climate, inundation frequency, substrates, moisture, salinity, hydrochemistry, herbivory, and predation (Verwey, 1954; Paine, 1974; Lubchenco, 1978; Semeniuk, 1983; Reise, 1985; Reise et al., 1994; Reise, 2000; Cronin et al., 2010).

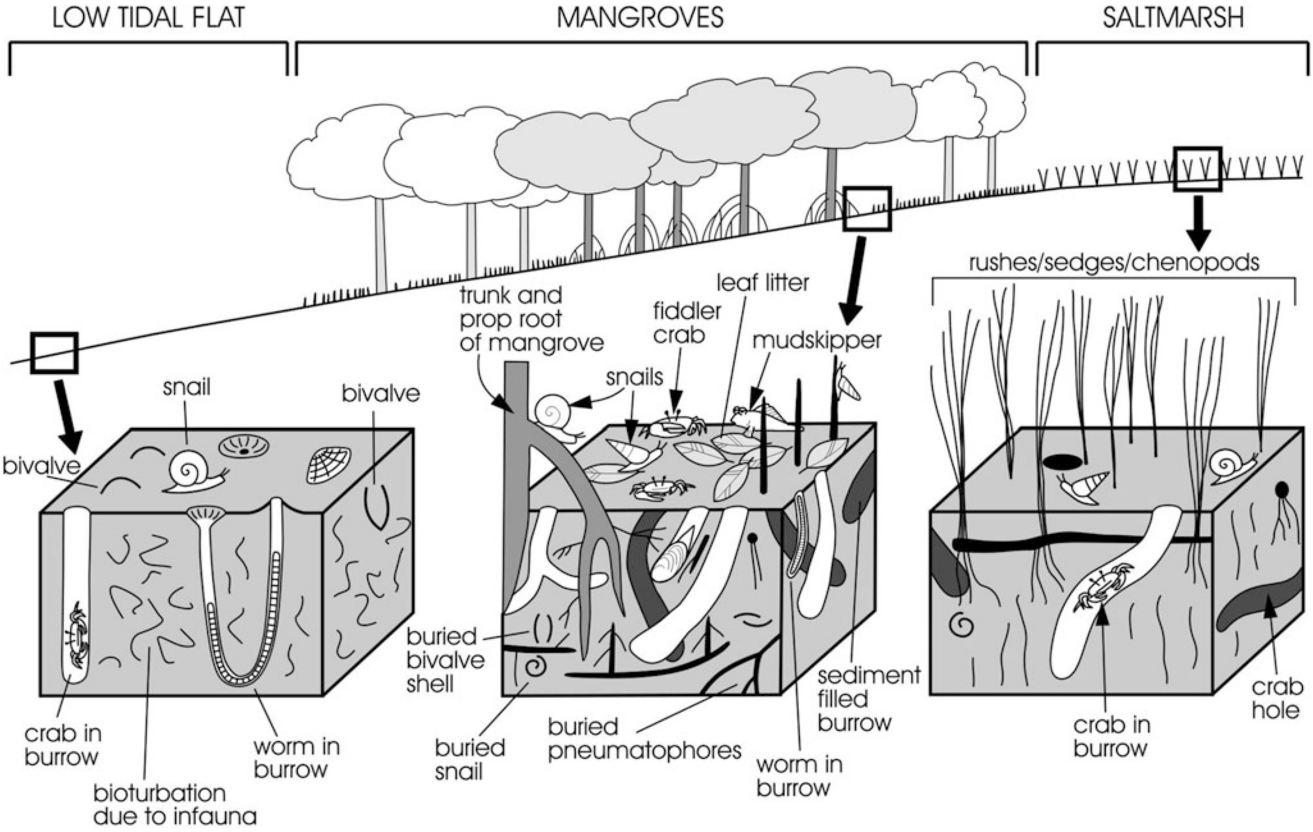
Biogeography and climate are major determinants of the composition of biota of tidal flats in estuaries. The range of biota includes mangroves and their associated fauna in tropical and subtropical environments, rushes, sedges, samphire, and other saltmarsh plants and their associated fauna in tropical, subtropical, and temperate environments, with different species in the various biogeographic regions (Tomlinson, 1986; Adams, 1990; Pen et al., 2000; Cresswell and Semeniuk, 2011), and various crustacean-polychaete-mollusc assemblages in tropical, subtropical, and temperate environments, again, with different species in the various biogeographic regions (Wolff, 1983; White, 1989; Semeniuk et al., 2000; Brearley, 2005). For instance, western hemisphere mangroves have different species than eastern hemisphere mangroves (Tomlinson, 1986), and their composition, structure, and physiognomy change with climate and latitude (Tomlinson, 1986; Cresswell and Semeniuk, 2011; Semeniuk and Cresswell, 2013).

Estuarine tidal flat biota comprise plants, resident fauna, invading and temporary fauna, and microbiota. The plants of tidal flats can be categorized into four main groups, viz., mangroves, saltmarsh, seagrass, and algae. Mangroves occupy the interval between MSL and MHWS but, in more humid and tropical climates, occupy the interval between MSL and EHWS. Saltmarsh occupies the interval between MSL and EHWS, but depending on climate, freshwater seepage, and tidal range, saltmarsh can more narrowly occupy the interval between MHWS and EHWS. Seagrass and algae generally occur in lower parts of the tidal flat and extend to subtidal zones.

Within a given biogeographic region, the composition of tidal flat biota, in site-specific locations, is related to tidal zones (inundation frequency), substrate type, substrate moisture, and salinity, commonly resulting in species zonation (Bridgewater, 1975; Wolff, 1983; Semeniuk, 1985; White, 1989; Pennings and Callaway, 1992; Emery et al., 2001; Pennings et al., 2005; Silvestri et al., 2005; and Semeniuk and Cresswell, this volume, see *Species Zonation*). Mangrove and saltmarsh emergent vegetation conspicuously exhibit species zonation (Semeniuk et al., 1978; Tomlinson, 1986; Adam, 1990; Cresswell and Semeniuk 2011).

There is a specialized and often unique fauna associated with mangroves, saltmarsh, and seagrass. For instance, in mangroves, there is a biogeography-specific and climate-specific mangrove-associated infauna of crustacea and polychaetes; an epifauna of crustacea (crabs, barnacles), insects, and vertebrates; and a canopy fauna of avifauna, reptiles, insects, and mammals (bats) (MacNae, 1968; Semeniuk et al., 1978; Hutchins and Saenger, 1987; Johnstone, 1990; Kathiresan and Bingham, 2001). Similarly in saltmarshes, there is a biogeography-specific and climate-specific saltmarsh-associated fauna, but also a partitioning of fauna relative to position of tidal levels and salinity. Adam (1990) uses several approaches to categorizing saltmarsh fauna: taxonomically, ecologically (e.g., marine, freshwater, and terrestrial affinities), trophically, subhabitat occupied, and residence status. For instance, for a given biogeographic and climate setting, saltmarsh may have an infauna of crustacea, other arthropods, molluscs, polychaetes, and meiofauna; an epifauna of crustacea, other arthropods, molluscs, insects, and vertebrates; and a foliage-associated fauna of avifauna, reptiles, mammals, gastropods, and insects (Adam, 1990).

The resident fauna of tidal flats outside of the zone of mangroves and saltmarsh, as macrofauna, can be diverse and complex in composition and its internal dynamics. Examples of compositional variability and complex dynamics are provided by Beukema (1976, 1981), Virnstein (1977), Bell and Coull (1978), Quammen (1982), Reise (1982, 1991, 2000), Ambrose (1984), Bottom (1984), Tsuchiya and Nishihira (1986), Dittmann (1990, 1995, 1996, 2000, 2002), Nehls and Thiel (1993), Jones et al. (1994), Thiel and Dernedde (1994), Vargas (1988), Ragnarsson and Raffaelli (1999), Attrill et al. (2001), Dittmann and Vargas (2001), Hagberg et al. (2004), Reichert and Buchholz (2006), Büttger et al. (2008), and Buschbaum et al. (2009). In the first instance, such fauna can be categorized as epifauna



Tidal Flat, Figure 5 Diagram showing blocks of sediment with different types of biota commensurate with substrate type and tidal level setting.

(inhabiting the tidal flat surface; this includes grazing gastropods, scavenging echinoderms, sessile mussels, encrusting oysters), infauna (inhabiting the tidal flat subsurface in structured burrows, or cavities, or simply buried; this includes bivalves, polychaetes, echinoderms, and crustacea; endobenthic is a term also applied to infauna), or alternating infauna/epifauna (inhabiting the subsurface but emerging periodically onto the surface for feeding or breeding; this includes crabs, goby fish, some polychaetes, and some gastropods, e.g., Unno and Semeniuk, 2011). Again, these assemblages are biogeography specific and climate specific in composition and structure but also substrate specific in that low tidal sand flat fauna can be markedly different from low tidal mud flat fauna for a given biogeographic and climate setting. A summary diagram of the complex associations of biota on unvegetated low tidal flats, in mangrove zones, and in saltmarsh is presented in Figure 5. A summary of the distribution, zoning, and environment-specific burrow structures (ichno-structures) developed across tidal flats by various fauna, and their development of structures is provided by Reineck and Singh (1980).

Microbiota of tidal flats includes diatoms, foraminifera, amoeba, bacteria, fungi, and small-scaled invertebrates such as annelids, nematodes, turbellarians, ostracods, crustacean larvae, insect larvae, and juvenile molluscs, among others (Higgins and Thiel, 1988). The small-sized fauna that lives interstitial to sand in sandy habitats and that generally passes through a 1 mm mesh (but retained by a 30–45 μm mesh) in sandy and muddy habitats is termed meiofauna. Meiofauna and other microbiota are abundant in tidal flat sediments as epibiota and endobiota, often zonally distributed, and may be vertically mobile with the rise and fall of the tide and groundwater under tidal flats (Wood, 1967; Happey-Wood and Jones, 1988; Brotas and Plante-Cuny, 1998; Hortona et al., 1999; Haslett, 2001; Mitbavkar and Anil, 2002, 2004; Coull, 2009; Ghosh, 2012; Hankin et al. (2012). They are commonly the food source for grazing and sediment-foraging invertebrates.

A special resident fauna on tidal flats includes some species of molluscs and worms (mussels, oysters, serpulid worms) that form dense colonies resulting in skeletal structures that cover the tidal flats as reefal biostromes or locally emerge as reefal bioherms (skeletal buildup terms from Nelson et al., 1962). Oysters and mussels typically can form sheetlike skeletal structures (or reefal biostromes) that cover the tidal flat in patches or ribbons

(Kuenen, 1942; van Straaten and Kuenen, 1957; Bosence, 1979; Dittmann, 1990; Nehls and Thiel, 1993; Eisma, 1998). Where hard surfaces are developed (such as mussel beds, oyster reefs, serpulid reefs, shell-lag pavements, or scattered shells on the sediment surface), encrusting organisms and (micro-scaled) hard-surface communities such as oysters, hydrozoans, anthozoans, bryozoans, and algae can develop (Albrecht, 1998).

The invading or temporary fauna of tidal flats, i.e., fauna that is not permanently resident on the tidal flat, include fish (including rays), crabs, reptiles, and octopus that invade the tidal flat on a high tide to feed; birds, reptiles, and mammals that invade the tidal flat on a low tide to feed; and the birds that use the saltmarsh vegetation and mangrove canopy for nesting, feeding, roosting, and loafing.

Figure 6 illustrates in generalized form the contrast between biota across a tropical macrotidal sandy to muddy tidal flat and that of a tropical macrotidal wholly muddy tidal flat, and the contrast between biota in various temperate-climate mesotidal and microtidal sandy tidal flats. Figure 7 illustrates various habitats and ecological/biological aspects of tidal flats. Figure 8 shows some of the biological aspects of the tidal flats.

Tidal flats as ecological systems – complexity, and trophic and ecologic function

Tidal flats are complex systems – from the large scale of geomorphology, habitats, ecosystems, geochemistry, hydrology, and hydrochemistry to the smaller scales of autoecology and microbial processes. Occurring in the coastal zone, as the low-gradient interface between ocean, land, atmosphere, and groundwater, they support a diversity of biota and a diversity of physical, chemical, and biological processes and, as such, function as a storage system for living and decaying biomass and for biochemical/geochemical products. As a coastal system bridging the environments of ocean and land, and in estuarine settings bridging the environments of freshwater and marine water, tidal flats also provide a multiplicity of ecological functions. As a major zone of primary and secondary productivity, they are an important food source for marine and terrestrial animals.

In the first instance, there is primary production by benthic plant life such as mangroves, saltmarsh vegetation, seagrasses, algae, and diatoms and, during the high tide, primary production in the water column by phytoplankton. Plant biomass is then cycled through the tidal ecosystem by herbivory and/or nectivory by insects, gastropods, crabs, turtles, bats, fish, and mammals, leaf litter fall and death and decay of plant matter, and filter feeding by bivalves and other invertebrates. Plant material in decayed or comminuted form accumulates as detritus where it is either ingested by detritus feeders (gastropods, polychaetes, crustacea) or transformed by microbiota to microbial slime and grazed by gastropods, polychaetes, and crustacea, or (if mobilized into the water column) extracted by filter feeders, or where it is incorporated on and into the sediments becomes involved in complex geochemical and biochemical transformations and recycled by microbiota as part of the nitrogen, sulfur, phosphorus, and carbon cycle (see "*Mineralization*") and transformed into inorganic products. Following incorporation of biomass and detritus (in various stages of decay) into herbivores, nectivores, and filter feeders, these primary consumers are preyed on by predators such as gastropods, echinoderms, nektonic fish, demersal fish, stingrays, octopus, and terrestrial invaders such as snakes, other reptiles, mammals, and wading birds.

As such, tidal flats are major feeding grounds for nekton and demersal fish on the high tide and major feeding grounds for reptiles, mammals, and avifauna on the low tide. Figure 9 diagrammatically shows the ecological function of mangroves on a tidal flat, with blocks of sediment showing different types of biota and their activity commensurate with substrate type and tidal level setting, and the invasion of terrestrial animals on the low tide and marine animals on the high tide to feed on the resident biota of the tidal flat.

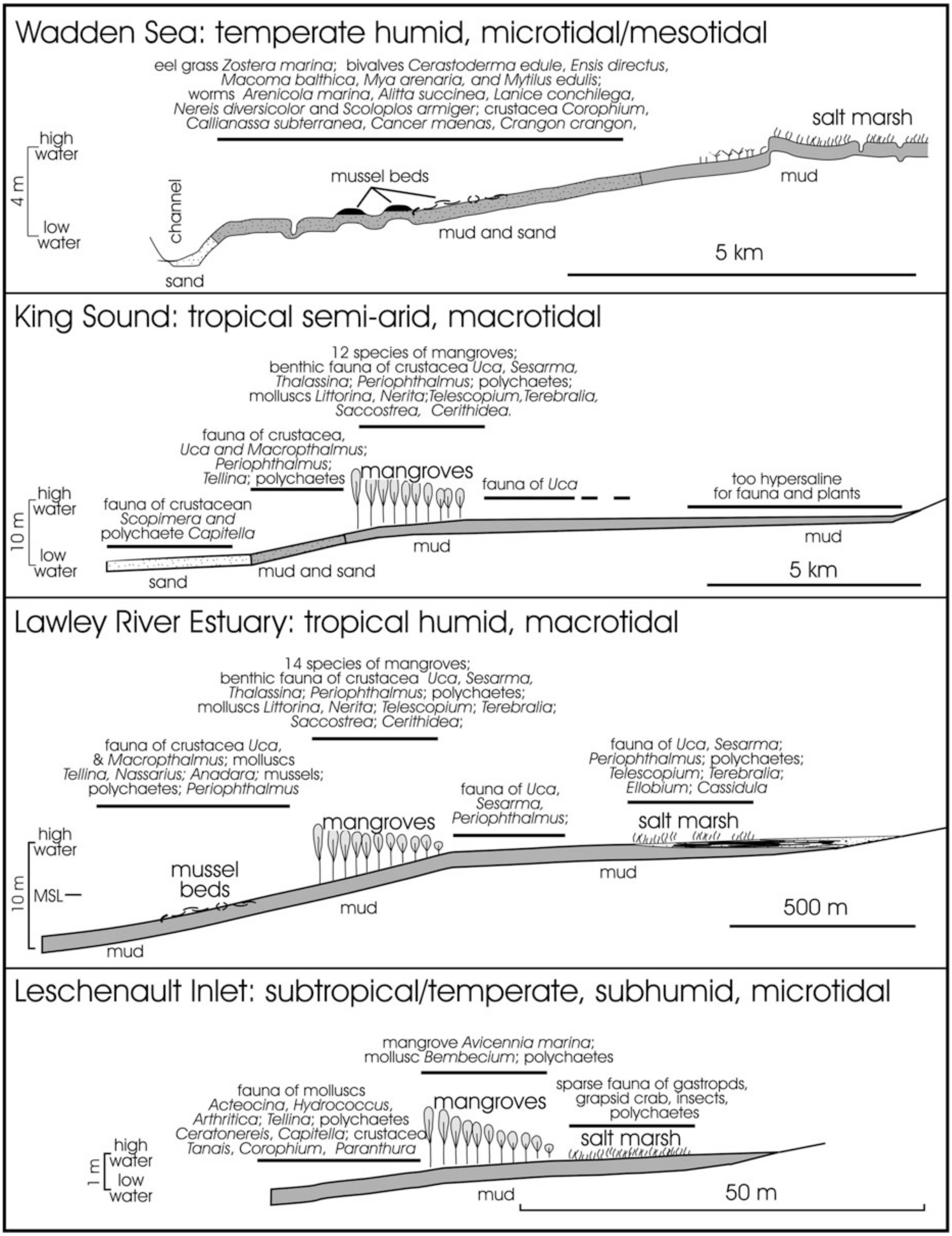
Examples specifically of bird life utilizing the tidal flats, their dependence on the ecosystem and effects on the ecosystem, and their nesting in mangroves and saltmarsh are provided by Semeniuk et al. (1978), Goss-Custard (1980), Zwarts (1981), Smit (1984), Johnstone (1990), Dann (1991), Beukema et al. (1993, 2010), Nehls and Tiedemann (1993), Piersma et al. (1993), Ens et al. (1994), Goss-Custard et al. (1996), Backwell et al. (1998), Raines et al. (2000), Connolly and Colwell (2005), Granadeiro et al. (2007), and Spalding et al. (2010).

Tidal flats and estuarine shallow waters also are considered to be nurseries and sanctuaries for fauna (McHugh, 1967; Blaber and Blaber, 1980; Staples, 1980; Haedrich, 1983; Boehlert and Mundy, 1988; Beukema, 1992; Maes et al., 1998; Morrison et al., 2002; Francis et al., 2005; Reise et al., 2010) – for contrasting discussions of the importance of shallow-water habitats for fish nurseries, see Able (2005), Baker and Sheaves (2005, 2007), and Johnston et al. (2007). While shallow waters are considered to be nurseries for fish and juvenile crabs and shrimps, nurseries also are specifically provided on tidal flats themselves by pools of water that are left stranded on the low tide. During periods of recruitment, the numerous small water pools on uneven surfaces of open tidal flats, in mangroves, and in saltmarsh are abundant with small fish. Sanctuaries are provided in the mangroves, saltmarsh, and seagrass.

Tidal flats provide variable habitats for organisms related to sediment types, tidal levels, and hydrochemistry and, as such, for a given biogeography and climate setting, provide the basis for biodiversity.

Summary

Tidal flats are tidally exposed surfaces, underlain by a variety of substrates, and variably vegetated depending on climate and tidal level. The entire surface between



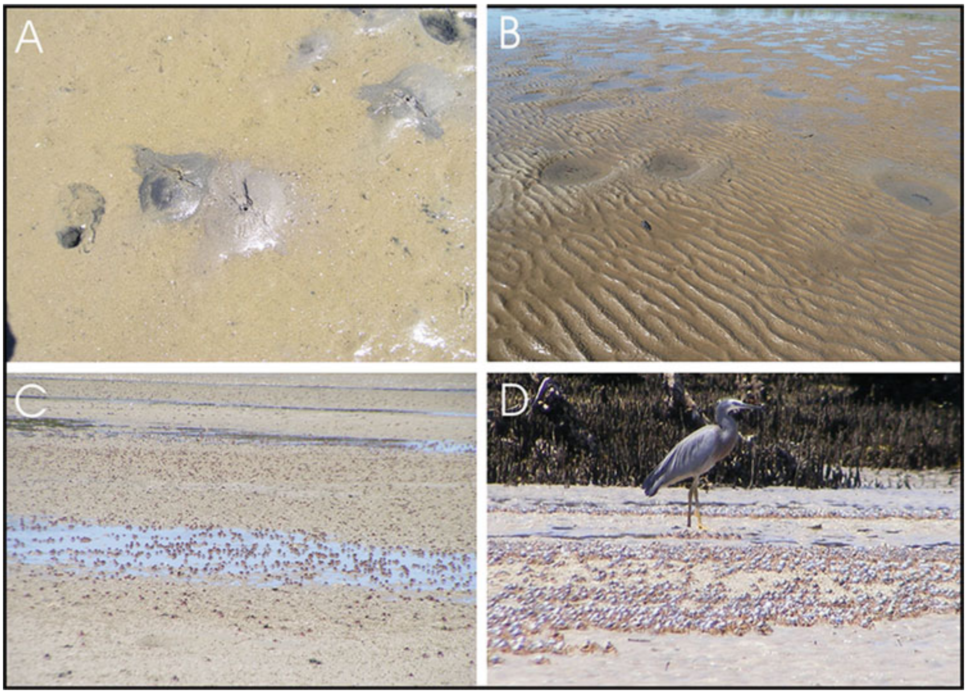
Tidal Flat, Figure 6 Profiles of some typical estuarine tidal flats showing substrate types and generalized and simplified composition of biota (mangrove, saltmarsh, invertebrates, mussels). Information on fauna in Wadden Sea from Wolff (1983); in the Lawley River Estuary and King Sound from Semeniuk (1981), Wells and Slack-Smith (1981), and Semeniuk (unpublished data); and in Leschenault Inlet Estuary from Semeniuk and Wurm (2000), Semeniuk (2000), and Dürr and Semeniuk (2000).

Tidal Flat, Figure 7 Various tidal flats in estuaries and some of their ecological/biological aspects. (**a**) Sandy low tidal flat, with low-relief shore-parallel shoals and depressions (small-scale habitats), in a northern Kimberley estuary, northwestern Australia. (**b**) Zoned mangroves on a muddy tidal flat in the northern Cambridge Gulf estuary, northwestern Australia; landward of the mangroves is a vegetation-free saline muddy flat (salt flat); spit emanates from the rocky spur and crosses the salt flat. (**c**) Zoned mangroves on a muddy tidal flat in northern King Sound, northwestern Australia; seaward of the mangroves is a moderately sloping mud flat, and landward is a vegetation-free muddy salt flat; also on the salt flat are sand bodies (cheniers). (**d**) Saltmarsh along shore of the Leschenault Inlet Estuary, southwestern Australia, showing three vegetation zones – shoreline rush in immediate foreground, samphire in foreground to middle ground (alternating bands of *pink*, *green*, and *grey*), and samphire in the background (*pink*); *Avicennia marina* shrub occurs in seaward saltmarsh zone. (**e**) Low tidal muddy sand flat adjoining a sloping sandy and shelly beach, Tasmania, southeastern Australia. (**f**) Low tidal muddy flat adjoining a mud-floored mangrove fringe, Botany Bay, southeastern Australia; the mud flat is pocked with worm burrows, crab burrows, and fish-feeding excavations. (**g**) Slightly muddy sand low tidal flat, Tasmania, southeastern Australia; soldier crabs swarming and feeding on surface. (**h**) Low tidal sandy flat with soldier crab feeding pellets on the surface, Tasmania, southeastern Australia; in background is a shore-parallel low-relief shoal; in foreground, the shiny surface is due to a thin film of mud giving the impression the tidal flat is a mud flat.

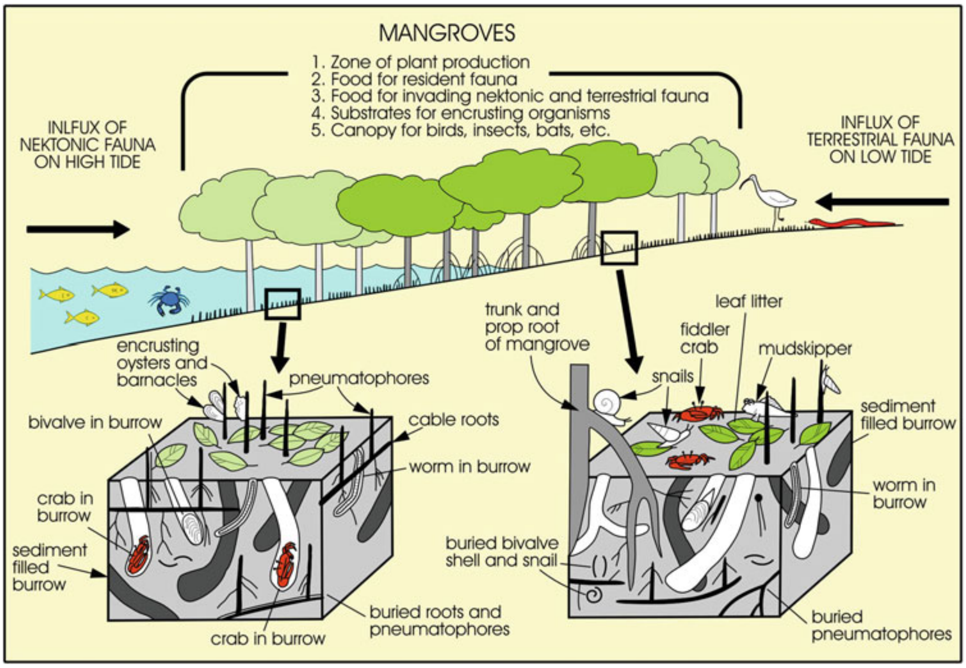
ELWS and EHWS is the tidal zone, and any low-gradient surface between these levels is a tidal flat. Tidal flats are not always simple uniformly sloping surfaces; they often have variation as determined by hydrodynamic setting, e.g., low hummocks of sand on mud flats, low relief cheniers, or tidal creeks, as well as variable substrates. These smaller-scale variations influence the development of small-scale habitats and the distribution of biota across the tidal flat. In providing variable habitats for organisms related to sediment types, tidal levels, and hydrochemistry, for a given biogeography and climate setting, tidal flats form the foundation for coastal biodiversity.

Tidal flats occur in varied geomorphic and sedimentologic settings within an estuary; these include platforms bordering estuary margins, tidal parts of flood-tidal deltas, tidal parts of intra-estuarine riverine deltas, the crests and flanks of middle-of-estuary shoals, and the protected lagoons leeward of spits. Where sand and mud are available, the sediments commonly are partitioned across the tidal flat because of hydrodynamic processes, resulting in depth-related mud flat grading to sand flats.

When exposed at low tide, tidal flats are subject to evaporation and transpiration, and a salinity gradient from seaward to landward is developed due to the interacting



Tidal Flat, Figure 8 Biology of some tidal flats of estuaries in southeastern Australia. (**a**) Polychaete workings of sediment (low-relief conical mounds) and burrows on a low tidal flat, Tasmania. (**b**) Fish and stingray feeding excavations on a rippled low tidal sandy flat, Tasmania. (**c**) Slightly muddy sand low tidal flat with shallow pools, Tasmania; swarm of soldier crabs on the surface. (**d**) Swarm of soldier crabs on low tidal flat and white-faced heron (*Egretta novaehollandiae*) feeding on the crabs, Botany Bay, southeastern Australia.



Tidal Flat, Figure 9 Diagram showing ecological function of mangroves on a tidal flat with blocks of sediment showing different types of biota commensurate with substrate type and tidal level setting (modified after Semeniuk et al., 1978).

effects of inundation, evaporation, transpiration, and sediment types. The increasing salinity can be locally diluted by freshwater seepage from the margin of the hinterland, beaches, and spits and cheniers.

Estuarine tidal flat biota comprise plants, resident fauna, invading and temporary fauna, and microbiota. At the overarching scale, biogeography and climate are major determinants of the composition of tidal flat biota. Tidal flat biota includes mangroves and their associated fauna in tropical and subtropical climates, and rushes, sedges, samphire, and other saltmarsh plants and their associated fauna in tropical, subtropical, and temperate climates. Mangrove and saltmarsh emergent vegetation conspicuously exhibits species zonation. A range of mollusc, crustacean, and polychaete fauna occurs in tidal flat habitats, the composition of which is related to tidal zones, substrate type, substrate moisture, and salinity, and commonly results in species zonation. The resident fauna of tidal flats can be diverse and complex in composition and its internal population dynamics and ecological dynamics. It can be categorized as epifauna, infauna, or alternating infauna/epifauna and is biogeography specific, climate specific, and substrate specific in composition and structure. The invading or temporary fauna of tidal flats are variable in terms of taxonomic group; they include fish (including rays), crabs, reptiles, and octopus that invade tidal flats on a high tide to feed; birds, reptiles, and mammals that invade tidal flats on a low tide to feed; and the birds that use the saltmarsh vegetation and mangrove canopy for a variety of activities.

A special resident fauna on tidal flats of mussels, or oysters, or serpulid worms can form dense colonies resulting in skeletal structures that cover tidal flats as reefal biostromes or locally emerge as reefal bioherms. Where hard surfaces are developed, encrusting organisms and (micro-scaled) hard-surface communities of oysters, hydrozoans, anthozoans, bryozoans, and algae can develop.

Microbiota of tidal flats includes diatoms, foraminifera, amoebae, bacteria, fungi, and small-scaled invertebrates that are abundant in tidal flat sediments, often zonally distributed, and vertically mobile with the fluctuation of the tide and groundwater of tidal flats. They are commonly the food source for grazing and sediment foraging invertebrates.

Ecologically, tidal flats are complex systems, even if substrates appear relatively uniform (e.g., wholly muddy tidal flats or wholly sandy tidal flats). Complexity is underpinned by inundation gradients and salinity which will determine zonation and hence microecology processes. Where substrates are graded, or where there is variable low-relief geomorphic expression, and where freshwater is discharging into the tidal flat, the complexity of tidal flats increases. Occurring in the coastal zone, as a - low-gradient interface between ocean, land, atmosphere, and groundwater, they support a diversity of biota and a diversity of physical, chemical, and biological processes, and they function as a storage system for living and decaying biomass and for biochemical/geochemical products. As coastal systems bridging the environments of ocean and land and in estuarine settings bridging freshwater and marine water, tidal flats also provide a multiplicity of ecological functions. As major zones of primary and secondary productivity, they are important food sources for marine and terrestrial animals.

On tidal flats, there is primary production carried out by benthic plant life such as mangroves, saltmarsh vegetation, seagrasses, algae, and diatoms and, during the high tide in the water column, by phytoplankton. Plant biomass is cycled though the ecosystem by herbivory and/or nectivory, leaf litter fall, decay of plant matter, and filter feeding by invertebrates. Following incorporation of biomass and detritus by primary consumers, predators carry the biomass into higher trophic levels. Tidal flats are major feeding grounds for nekton and demersal fish on the high tide and for reptiles, mammals, and avifauna on the low tide. Tidal flats, and in particular mangroves, saltmarsh, and seagrass, also provide nurseries and sanctuaries for fauna.

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Cross-references

Estuarine Deltaic Wetlands
Mangroves
Saltmarshes
Sandflat
Soldier Crabs (Mictyridae)
Tidal Flat Salinity Gradient

TIDAL FLAT SALINITY GRADIENT

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Definition

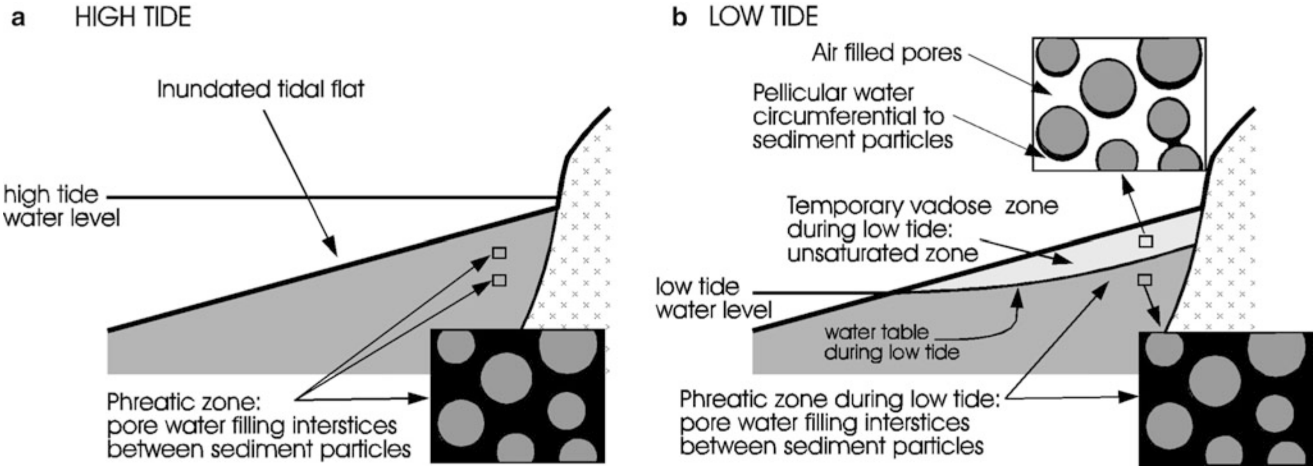
A tidal flat salinity gradient is the systematic or progressive across-tidal-flat change in salinity in groundwater (as pore water) and pellicular water due to the interacting effects of inundation, evaporation, transpiration, sediment types, and freshwater seepage.

Tidal flat salinity

During low tide, when tidal flats are exposed, water under or within a tidal flat is subject to evaporation and transpiration, resulting in its salinization, or to dilution by freshwater seepage. During a low tide, there are two sub-environments where tidal flat water resides (Figure 1):

1. in a shallow groundwater that is contiguous with the open waters and whose water table falls and rises with the rhythm of the tide; at slack-water low tide, this water table can be ~2–10 cm deep on low-gradient muddy tidal flats and sandy tidal flats or located up to 30 cm depth on more steeply inclined sandy tidal flats; at extremes, the water table can be at the surface on very low-gradient tidal flats; the groundwater, which also can be referred to as the phreatic zone, resides in the pore spaces of the sediments – this is tidal flat pore water.
2. as water films circumferential to sediment particles in the wetted but undersaturated sediment above the water table during low tide; this can be referred to as the temporary vadose zone – this is tidal flat pellicular water.

The salinity of tidal flat groundwater and pellicular water varies spatially and temporally depending on the location on the tidal flat relative to mean sea level (MSL), the stage of the tide, the time length of exposure of the tidal flat within a tidal cycle or between spring tides and neap tide phases, and the salinity of the source water. The salinity of the tidal flat groundwater tends to be relatively constant from tide to tide during a given season and can even be constant over the entire year. However, depending on the style of recharge into the pore waters by flooding surface tides and a water table rising with a tide, the salinity of the groundwater of estuarine tidal flats can change seasonally. Although the water table of the groundwater under tidal flats is shallow, it is separated from the surface by a temporary vadose zone, and at low tide generally, it is not in direct contact with the agencies of solar-induced evaporation and wind shear. During a flooding tide, the groundwater by rising into the temporary vadose zone salinized during the previous low-tide period redissolves salt, hence raising its salinity. Plants on the tidal flat, while clearly utilizing pellicular water in



Tidal Flat Salinity Gradient, Figure 1 (**a**) The characteristics of the water residing under tidal flats during high and low tide. The components of the tidal flat hydrology during low tide are annotated in inset (**b**).

the temporary vadose zone, also utilize waters of the phreatic zone. Transpiration by tidal flat vegetation increases the salinity of the pellicular water and the groundwater.

The temporary vadose zone is wetted and dried on a tidal cycle and is subject to more direct evaporative processes, causing the salinity of its pellicular water to be more variable, to fluctuate over a tidal cycle, and to vary from spring tide to neap tide and between the seasons.

Controls of tidal flat salinity

There are a number of factors that control local tidal flat salinity and tidal flat salinity gradients. These are as follows: (1) recharge of tidal flat pore waters and pellicular waters by tidal inundation, (2) decreasing inundation of the tidal flat upslope and increasing exposure to evaporation by solar radiation and wind, (3) transpiration, (4) the tidal range and nature of the tide, (5) the tidal flat slope, (6) sediment types, (7) shoaling sediment bodies, (8) homogeneity or heterogeneity of the tidal flat stratigraphy, and (9) freshwater seepage onto tidal flats from the hinterland.

Specifically for tidal flats within estuaries, the salinity of the open estuarine waters that daily recharge/inundate the tidal flats and recharge the pore waters and pellicular waters is a large factor in influencing tidal flat salinity as this determines the starting salinity of the tidal flats that will be subject to evaporative concentration.

Tidal flats facing open estuarine waters that are near the marine realm and predominantly of marine salinity will be recharged by waters of salinity ~35 ppt, and evaporative concentration will begin from that salinity. A fourfold increase in salinity in higher parts of such tidal flats may result in a salinity of ~140 ppt. Tidal flats facing open estuarine waters that are predominantly brackish (for instance, in mid-estuarine locations) will be recharged by waters with, say, ~10–20 ppt salinity; a fourfold increase in salinity in higher parts of the tidal flat in such situations may result in a salinity of ~40–80 ppt. Tidal flats facing open estuarine waters that are predominantly freshwater (for instance, in riverine and near-riverine locations in the upper estuary) will be recharged by waters of much lower salinity, and a fourfold increase in salinity in higher parts of the tidal flat may result in a salinity still in the freshwater field or lower brackish water field. In each case, the tidal flat pore water and pellicular water will begin their evaporative concentration from the level of the salinity of the open waters that recharge the tidal flats.

The topography of the tidal flat sloping from low tide to high tide is a gradient of decreasing inundation. As such, the progressively higher parts of a tidal flat are exposed to increasing evaporation. The effects of solar radiation and wind are direct, as they interface with large and extensive wet or moist tidal flat surfaces. As water is evaporated from the tidal flat surface (thereby concentrating salts or even precipitating salts) and surface sediment begins to dry, capillary action can deliver pellicular water to the surface from the shallow water table underlying the tidal flat either incorporating pellicular water with concentrated salts or dissolving the precipitated salts. Both processes increase salinity.

Transpiration by tidal flat vegetation such as mangroves, saltmarsh, samphires, and seagrass also results in moisture loss from tidal flats and in the concentration of salts. Vegetation type on the tidal flat and its transpiration rate determine the extent of moisture loss and the increase in salinity.

The tidal range and the nature of tide (whether it is diurnal, semidiurnal, or mixed) determine the period that a tidal flat is exposed and the area of exposure. Microtidal coasts (say, with tidal ranges <0.5 m) tend to have relatively narrow tidal flats and the exposure gradient between low and high water results in relatively narrow shore-parallel zones of differing exposure indices that increase towards the shore. Macrotidal coasts (say, with tidal ranges >4 m, and locally over 10 m) tend to have relatively wide tidal flats, particularly if the tidal flats have a very low topographic gradient, and the exposure between low and high water results in wide shore-parallel zones within a tidal flat of differing exposure indices increasing towards the shore. In this latter situation, the gradient of salinity of the tidal flat is markedly evident between MSL and high water and follows the shore-parallel exposure indices.

In combination with sediment type, the tidal flat slope determines how rapidly during a low tide the tidal flat drains by vertical infiltration or by direct runoff. Tidal flat slope also determine how rapidly tidal flat water changes from phreatic (with its water table at, or close to the tidal flat surface) to vadose (the wetted sediment above a falling water table at or close to the tidal flat surface). Topographically very low-gradient tidal flats, particularly if they are muddy and not sandy, drain water relatively slowly such that there is a longer residency time of pore water and pellicular water in the surface and hence a longer period for evaporation. Steeper flats, particularly if they are sandy, tend to drain water more quickly with shorter residency time for surface pore water and pellicular water and less time for direct evaporation.

Sediment types also determine how rapidly the tidal flat drains during low tide. Sandy substrates are more hydraulically transmissive than mud substrates and drain water more rapidly laterally and vertically. Because of their low transmissivity, surface layers of mud and muddy sand retain water for a longer period, are subject to evaporative processes for longer periods, and tend to salinize more rapidly than sand. While tidal flats that are composed wholly of mud or wholly of sand will manifest relatively uniform but different shore-parallel gradients of salinity in groundwater and pellicular water in response to inundation, evaporation and transpiration, tidal flat sediments that vary from sand to mud in cross-tidal-flat gradients, or mosaics of sand and mud will exhibit more complicated salinity patterns as the sediment types become additional controlling factors.

The shore-parallel zones of increasing groundwater and pellicular water salinity from MSL to high water in response to decreasing inundation and increasing evaporation and transpiration and their resultant salinity gradients can be disrupted by the hydrological changes effected by vertical sedimentary shoaling to levels above the high water mark and by discordant (or heterogeneous) stratigraphy such as sandbars and cheniers. In humid environments with excess rainfall, the sedimentary sequence may vertically shoal (accrete) to levels of high water and then be capped by peat (Coleman et al., 1970). Peat, sustained by rainfall, may continue to accrete vertically to form peat mounds to levels above high tide. Such peat mounds, being above the level of high tides, are replenished by rainfall and store freshwater as subsurface mounds which discharge into the adjoining tidal flats diluting and perturbating the pattern of tidal flat salinity. Similarly, a discordant body of topographically emergent sand on a muddy tidal flat (e.g., a sandy chenier that has emerged above high water) will not be hydrologically and hydrochemically influenced by flood-tide recharge but by rainfall. The chenier, on receiving rainfall, stores freshwater as a subsurface lens which, during times of elevated water tables after rain, can discharge into the surrounding tidal flat, perturbating the salinity gradient (i.e., the increase in salinity from MSL to high water on the surrounding tidal flat). The extent of the perturbation/dilution by the freshwater stored in a chenier on the salinity of the tidal flat depends on where the chenier is located. The most profound effect is by cheniers and their freshwater lenses on the higher parts of tidal flats where there is the most marked contrast between freshwater and the tidal flat salinity.

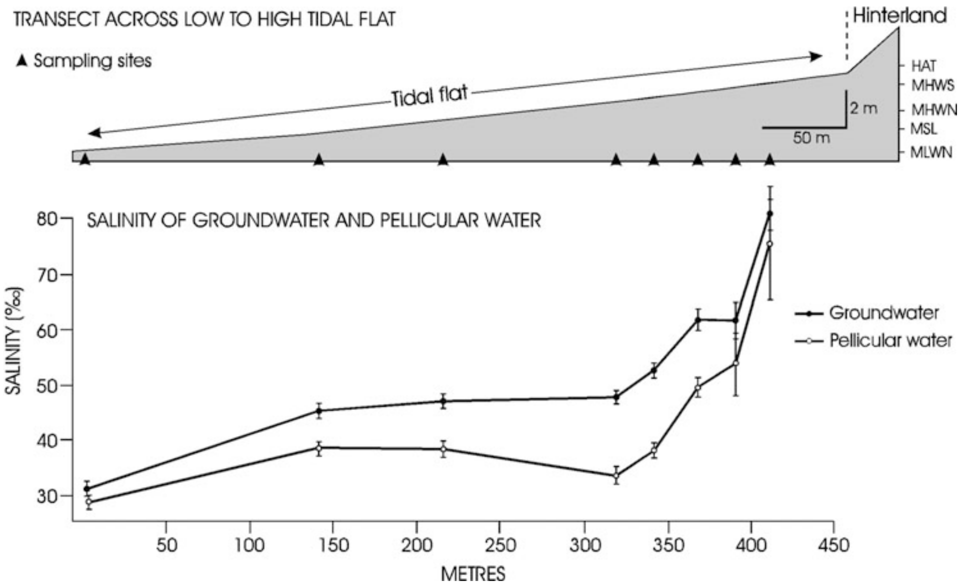
As such, tidal flats generally have a gradient of increasing salinity from low-tidal zones to high-tidal zones (Figure 2).

While there may be a general trend of increasing tidal flat salinity from low-tidal levels to the high-tide mark, this can be reversed by freshwater seepage along the margin of the hinterland. Freshwater seepage can manifest as discharge along the upper tidal flat surface as surface flows or springs or as subterranean flow along appropriate stratigraphic conduits or sheets.

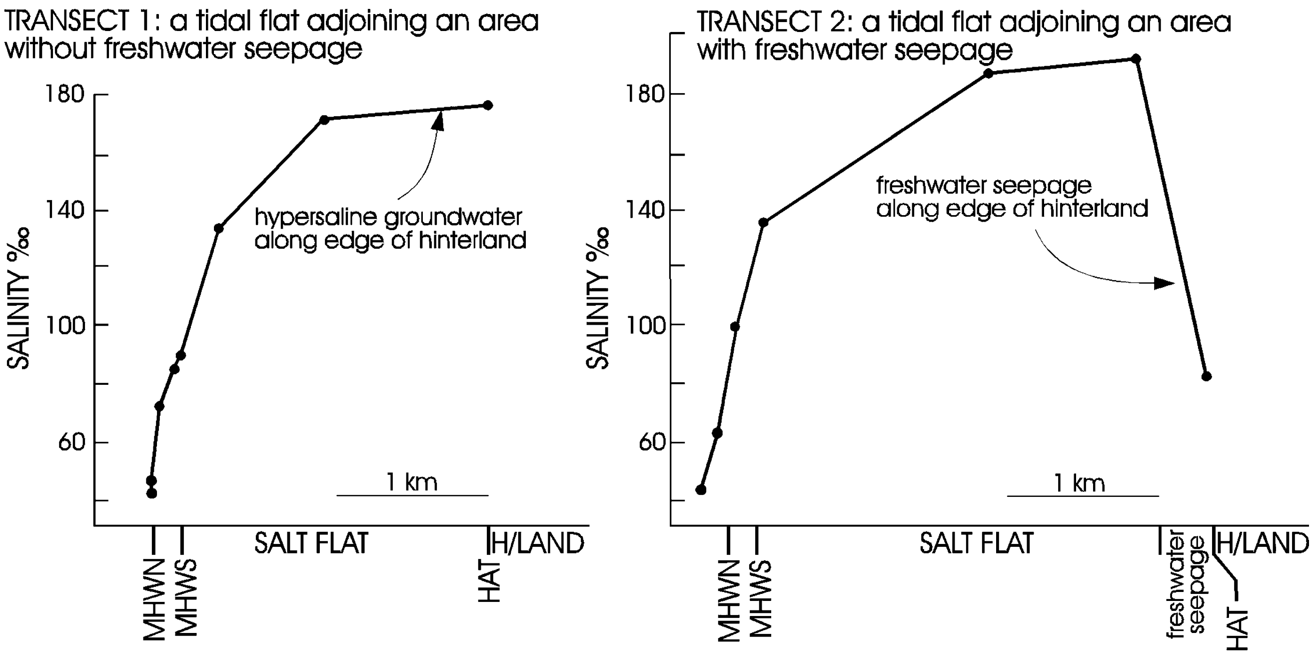
Tidal flat salinity gradients

Examples are provided of tidal flat salinity gradients in Figures 3 and 4 from a tropical macrotidal estuarine muddy tidal flat where rainfall is ~1,548 mm per annum and evaporation is ~2,800 mm per annum and from subtropical (near-temperate) microtidal estuarine sand-and-mud tidal flat where rainfall is ~880 mm per annum and evaporation is ~1,300 mm per annum (Figure 4). In both cases the open estuarine waters are near marine in salinity. In the tropical macrotidal estuarine example, the tidal flat salinity increases from 35 to 40 ppt at MSL to ~ >180 near the level of the highest astronomical tide (HAT). In the subtropical microtidal estuarine example, the tidal flat salinity increases from 35 ppt at MSL to ~60 ppt at mean high water spring tide (MHWS), but is diluted by freshwater seepage along the hinterland edge (see below). In areas of seasonal high rainfall, and where the tidal flats are underlain by sand and bordered by a hinterland of sand, the hinterland may be an aquifer storing and seasonally discharging freshwater by seepage into the tidal flat – while there is a salinity gradient from low-tidal to high-tidal level, it seasonally changes from saline to brackish in the higher parts of the tidal flat over the dry to wet season.

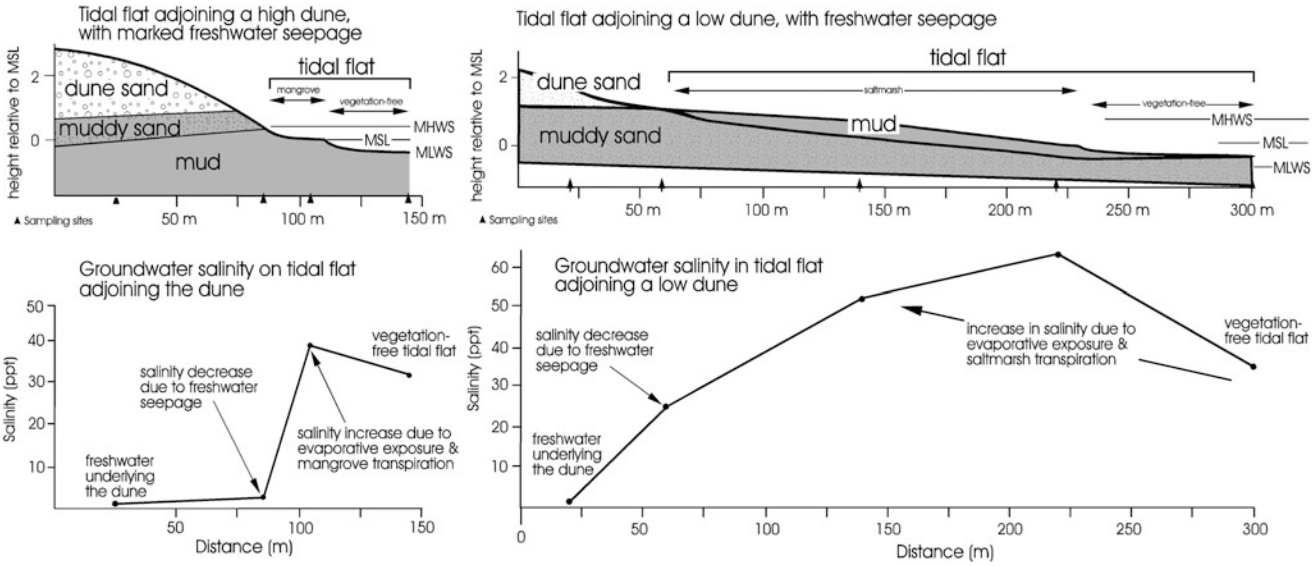
In estuaries where there is a strong seasonal to perennial delivery of freshwater by surface seepage or by subterranean seepage from supratidal environments (Semeniuk, 1983; Cresswell, 2000), the gradient of increasing salinity upslope can be locally reversed from hypersaline to



Tidal Flat Salinity Gradient, Figure 2 Salinity of groundwater and pellicular water across a tropical tidal flat in northwestern Australia. The salinity of the groundwater and pellicular increases from the low-tidal zone to the high-tidal zone, with the groundwater being of a slightly higher salinity than the pellicular water. There is no freshwater seepage in this location, and so the hypersalinity of the high-tidal flats is not diluted (compare with high-tidal salinity in Figures 3 and 4).



Tidal Flat Salinity Gradient, Figure 3 Salinity of groundwater across two transects of a tropical macrotidal tidal flat from the Lawley River delta of northwestern Australia (modified from Semeniuk, 1983). Along both transects, the salinity of the groundwater markedly increases from the level of mean high water neap tide (MHWN) to the level of the highest astronomical tide (HAT). Transect 1 is in an area with minimal freshwater seepage. Transect 2 is in an area with marked freshwater seepage, and as such, hypersalinity of groundwater is diluted along the hinterland edge.



Tidal Flat Salinity Gradient, Figure 4 Annotated profiles of the salinity of groundwater across two transects along a subtropical (near-temperate) microtidal tidal flat from the Leschenault Inlet Estuary, southwestern Australia (Modified from information in Cresswell (2000) and Semeniuk et al. (2000)). Along both transects, the salinity of the groundwater increases from the low-tidal level to the level of mean high water spring tide (MHWS). One transect borders a high dune, while the other borders a low dune. Both sites exhibit the effects of freshwater seepage from the dunes to the tidal flats.

brackish or to freshwater at the contact of the tidal flat with the supratidal zone. Examples of the effect of freshwater seepage on tidal flat salinity are shown in Figures 3 and 4.

Consequences of tidal flat salinity gradients

The main consequences of tidal flat salinity gradients are twofold: species zonation and mineral precipitation. The main effect of a tidal flat salinity gradient is on the occurrence and survivorship of various biota resulting in zonation of species across a tidal flat. The best examples of zonation are afforded by mangroves and salt marsh (MacNae, 1968; Tomlinson, 1986; Pennings et al., 2005; Silvestri et al., 2005) as they are conspicuous with different species visually distinct in terms of physiognomy, foliage color, and height. As such, their zonation is commonly evident as vegetation banding. The benthos of tidal flats also exhibit zonation, but here, since many such organisms are infaunal, the zonation needs to be brought out by sampling and mapping.

The other effect of a tidal flat salinity gradient is the progressive precipitation of minerals or the geochemical alteration of minerals. As pore waters and pellicular waters increase in salinity, precipitation of carbonate and sulfate minerals may occur (Logan, 1974). Calcite or aragonite may be precipitated at salinities >70 ppt, and gypsum may be precipitated at salinities >120 ppt. These minerals are precipitated interstitially (cementing the tidal flat sediments) or as nodules. Where there is freshwater seepage into the upper tidal flat, carbonate grain precipitation, carbonate grain dissolution, and (geochemical) carbonate grain alteration may occur. In the latter case, calcite grains may alter to dolomite (Scoffin, 1987; Semeniuk, 2010).

Discussion and summary

The prevailing pattern of tidal flat salinity across a tidal flat is one of increasing groundwater and pellicular water salinity upslope in response to decreasing inundation and increasing effects of evaporation and transpiration. For tidal flats recharged by marine water, the general trend is for the salinity to progressively increase from ~35 to 40 ppt between low water and MSL to in excess of 150 ppt at levels of high water. The magnitude and extent of increasing salinity upslope is determined by climate, which influences the amount of evaporation and precipitation, the salinity of the source water recharging the tidal flats, and the sediment types. Perturbations to a simple increasing upslope salinity gradient can result from mosaics of sediment types, such as sand patches amongst mud flats, with their potentially different water-holding capacity and drainage rates, and can result from the heterogeneous discordant emergent sediment bodies (such as cheniers and peat mounds) within tidal flat systems which can store and discharge freshwater onto/into the tidal flat resulting in local pockets of lower salinity. Freshwater seepage from the supratidal zone onto/into the tidal flat can have a marked influence on the salinity gradient and, through its diluting effect, can reverse the salinity gradient that was increasing from MSL to landward. Freshwater seepage affecting tidal flat salinity gradients is most marked in high-rainfall areas.

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Cross-references

Cheniers and Regressive Bedforms
Mangroves
Saltmarshes

TIDAL FRESHWATER HABITAT

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Synonyms

Tidal freshwater ecosystems; Tidal freshwater wetlands

Definition

Tidal freshwater habitat consists of wetland (See *Wetlands*) ecosystems, which are located at the far upstream end of estuaries and downstream from nontidal freshwater ecosystems. The salinity of tidal freshwater habitat is generally less than 0.5 parts per thousand (ppt). A variety of animal species, including fish, amphibians, reptiles, and particularly birds, live in tidal freshwater habitat permanently or for at least part of the year.

Description

Tidal freshwater habitat is a transitional habitat type found along the ecological continuum between tidal saline wetlands and nontidal ecosystems in both tropical and temperate coastal regions. Such habitat occupies the upstream reaches of estuaries and large coastal wetland complexes. For example, in the temperate zone, tidal freshwater habitat can be found upstream from salt marshes in estuaries along the West, Southeast, and Gulf coasts of the United States (Odum et al., 1984; Drexler et al., 2009b; Johnson et al., 2011). In the tropics, an example of tidal freshwater habitat is palm forest, which occurs upstream from mangrove ecosystems (Ewel, 2010).

Tidal freshwater habitat may contain forest, scrub-shrub, and herbaceous plant (marsh) communities. Examples of forest types (swamps) are bald cypress (*Taxodium distichum*) in the southeastern United States, dragon's blood (*Pterocarpus officinalis*) in the Caribbean basin, and nipa palm (*Nypa fruticans*) along coastlines and estuarine habitats of the Indian and Pacific Oceans (Rivera-Ocasio et al., 2007; Ellison et al., 2010; Dubenstein et al., 2013). Scrub-shrub wetlands typically contain a variety of woody vegetation generally less than 5 m in height. Examples of scrub-shrub plant species include smooth alder (*Alnus serrulata*), black willow (*Salix nigra*), and marsh elder (*Iva frutescens*) in the Southeast and Sitka willow (*S. sitchensis*), red osier dogwood (*Cornus stolonifera*), and hardhack steeplebush (*Spiraea douglasii*) in the Pacific Northwest of the United States (Thomas, 1984; Coulling, 2002). Herbaceous plant communities, particularly those found in the higher reaches of tidal freshwater marsh (Figure 1), tend to have greater plant diversity than forest or scrub-shrub communities. Typical high marsh species in the United States include annual plants, such as great ragweed (*Ambrosia trifida*), smooth beggartick (*Bidens laevis*), and knotweed (*Polygonum* spp.), and perennials, such as rice cutgrass (*Leersia oryzoides*), green arrow arum (*Peltandra virginica*), cattails (*Typha* spp.), giant cutgrass (*Zizaniopsis miliacea*), and bulrushes (*Schoenoplectus* spp.). In Europe, patch-forming perennials such as cattails (*Typha* spp.), bulrushes (*Schoenoplectus* spp.), reed canary grass (*Phalaris arundinacea*), valerian (*Valeriana officinalis*), and bur-reed (*Sparganium* spp.) tend to dominate the high marsh (Whigham et al., 2009; Barendregt and Swarth, 2013).

The physical conditions in tidal freshwater habitat reflect its transitional role in the landscape. The salinity of channels is usually less than 0.5 ppt. Tidal range (See *Tidal Ranges*) is typically from < 0.5 to 2 m and can potentially be greater upstream than downstream due to

Tidal Freshwater Habitat, Figure 1 An example of a tidal freshwater marsh in coastal South Carolina, USA.

constricting morphology that results in amplification of the tides (Mitsch and Gosselink, 2000). In some places, such as the Gulf Coast of the United States, wind-derived tides with irregular patterns of flooding and drying are much more important than lunar tides (Mitsch and Gosselink, 2000). Mature tidal freshwater habitat may contain thick, highly organic peat (See *Peat*) soils, which have accreted over hundreds or thousands of years (Drexler et al., 2009a, b).

A variety of animals, including fish, amphibians, reptiles, and waterfowl, live in tidal freshwater habitat permanently or for at least part of the year. With regard to fish, many spawn and spend their entire lives in channels within tidal freshwater habitat. Ictalurids (catfish), cyprinids (minnows, shiners, and carp), and centrarchids (sunfish, crappies, and bass) are three important families of such fishes. Many centrarchids, such as the largemouth bass (*Micropterus salmoides*) and sunfishes (*Lepomis* spp.), are important sport fish (Odum et al., 1984). Other estuarine fishes such as killifishes (*Fundulus* spp.) and juvenile naked gobies (*Gobiosoma bosci*) are part-time residents, using tidal freshwater habitat for a source of food or nursery grounds. A number of anadromous (See *Anadromous*) and semianadromous fishes (which live as adults in the ocean or lower estuary, respectively) travel through tidal freshwater habitat on their way to streams to spawn. The juveniles use tidal freshwater habitat as nursery grounds. In the Pacific Northwest, several threatened and endangered salmonid species, including the Upper Columbia River Spring Chinook salmon (*Oncorhynchus tshawytscha*), have been shown to rely on tidal freshwater habitat for this purpose (Johnson et al., 2011).

In addition to fish, a variety of amphibians and reptiles depend on tidal freshwater habitat. River turtles and water snakes are common inhabitants, but perhaps the most conspicuous residents are American alligators (*Alligator mississippiensis*), which are distributed throughout the southeastern United States (Mitsch and Gosselink, 2000). American crocodiles (*Crocodylus acutus*) may also be found in tidal freshwater habitat but are more likely to inhabit brackish and saline environments in their range, which extends from southern Florida to South America (Cherkiss et al., 2011). The much larger and broad-ranging saltwater crocodile (*Crocodylus porosus*) is distributed mainly throughout Southeast Asia and Australia and can be found in tidal freshwater habitat, particularly during the wet season (Campbell et al., 2013).

A number of mammals make their home in tidal freshwater habitat. Some of these inhabitants are particularly well adapted to wet environments such as beaver (*Castor canadensis)*, marsh rabbit (*Sylvilagus palustris*), marsh rice rat (*Oryzomys palustris*), river otter (*Lontra canadensis*), muskrat (*Ondatra zibethicus*), mink (*Neovison vison*), and nutria (*Myocastor coypus*; an invasive species (See *Invasive Species*) originally from South America) (Odum et al., 1984). Other mammals, including white-tailed deer (*Odocoileus virginianus*), opossum (*Didelphis virginiana*), and raccoon (*Procyon lotor*) may use tidal freshwater habitat to fulfill at least part of their food or habitat requirements (Odum et al., 1984).

The diversity of plants in tidal freshwater habitat provides a range of ecological niches (See *Ecological Niche*) highly suitable for a large number of resident and migratory bird species. In fact, because of the broad range of plant assemblages, tidal freshwater habitat supports some of the largest and most diverse bird populations of all wetland types. In a survey of tidal freshwater marshes, Odum et al. (1984) counted 280 species of birds in a broad range of sub-habitats. Shorebirds, including sandpipers, killdeer, woodcocks, dunlins, as well as rails, use low marsh and exposed mudflats. Seed-eating species such as red-winged blackbirds, sparrows, and bobolinks use the mid- and high marsh, which contains grasses and sedges. Herons, egrets, ibises, bitterns, and other wading birds depend on ponded areas and channels. The marsh plain and open water areas are prime habitat for ducks, geese, swans, and other waterfowl, which have been hunted by human populations for thousands of years (Barendregt and Swarth, 2013). Shrubs and trees are used by arboreal birds such as flycatchers and swallows. Tidal marsh habitat also provides hunting grounds for hawks, falcons, eagles, owls, and other birds of prey.

Today, tidal freshwater habitat is recognized for providing important ecosystem services such as biodiversity support and water purification (Barendregt and Swarth, 2013). Despite this, much tidal freshwater habitat has been transformed or lost, due to its close proximity to human populations. Extensive regions containing tidal freshwater habitat have been drained and reclaimed for agriculture (Penland and Ramsey, 1990; Ibanez et al., 1997; Drexler et al., 2009a). What remains has been impacted by nutrient loading, contaminants, and/or hydrologic diversions (Barendregt and Swarth, 2013). Sea-level rise presents a further threat to tidal freshwater habitat, particularly from changes in hydrology and increases in salinity, which may dramatically alter plant communities (Barendregt and Swarth, 2013). In suitable environments, wetland restoration may be used to reverse this trend and regain habitat values and ecosystem services provided by tidal freshwater habitat.

Summary

Tidal freshwater habitat consists of wetland ecosystems containing a wide variety of plant communities, a high diversity of birds, and plenty of other wildlife adapted to wet conditions. Such ecosystems are typically found at the far upstream end of estuaries. This position in the landscape makes tidal freshwater habitat vulnerable to localized impacts such as drainage, water diversion, and pollution as well as regional and global processes such as sea-level rise. In suitable environments, restoration of wetlands may be used to regain ecosystem services and habitat values provided by tidal freshwater habitat.

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Cross-references

Anadromous
Ecological Niche
Invasive Species
Peat
Tidal Ranges
Wetlands

TIDAL HYDRODYNAMICS

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Definition

Tidal hydrodynamics refers to the physics of vertical and horizontal water movement caused by the tidal forces of the moon and sun.

Introduction

In most estuaries, the dominant forces that cause the water to move are produced by the gravitational effects of the moon and sun on the oceans. These astronomical forces cause the water to move vertically (the tide) and horizontally (the tidal current), the entire phenomenon usually referred to as the *tides*. Tidal motion is actually the motion of extremely long waves generated in the deep ocean, which propagate over the shallower continental shelf and up into even shallower estuaries where their size is usually amplified.

Although it is astronomical factors that makes the tides so predictable, it is the hydrodynamics (i.e., physics of the water movement) that determines the size and timing of the tides. It is the hydrodynamics (determined by the dimensions of the ocean, continental shelves, bays, and rivers) that determines how large the tide range (height difference between high water and low water) will be and when the high and low waters will occur. It is the hydrodynamics that determines how fast the tidal currents will flow and when slack waters will occur, and it is the hydrodynamics that determines how significant the diurnal tidal signal will be compared with the usually dominant semidiurnal tidal signal.

Only the oceans are large enough for the tide-generating forces to directly produce a tide of significant size (see *Tides*, and also Parker, 2007). The tides in an estuary are forced at the estuary entrance by the tide wave from the deep ocean (having been modified to some degree by propagating over the shallower continental shelf). When we speak of the tidal hydrodynamics of estuaries, we are essentially speaking of shallow-water tides, which are significantly different than open-ocean tides in a number of ways, including having much larger tidal ranges and faster tidal currents. In shallow water the tidal wavelength is shorter, which is one factor leading to larger tidal amplitudes when the tide wave reflects from the head of an estuary. In estuaries, full and partial reflections of these very long tidal waves, the nearness of basin lengths to resonance for particular tidal frequencies, continuity effects, frictional damping, and advective/inertial effects all affect the tide and tidal currents. In shallow water the hydrodynamics also transfers tidal energy, through various nonlinear processes, to new frequencies. These new, so-called shallow-water tidal constituents can be larger than many of the astronomically generated tidal constituents. Some shallow-water constituents (called overtides) are higher harmonics of particular astronomical constituents, which distort the simple sinusoidal shape of tide and tidal current curves that are found in the open ocean, often causing asymmetry in the tidal cycle. These same nonlinear processes also lead to interactions between the tide and other nontidal phenomena such as wind-induced changes (e.g., storm surges) and river discharge.

Brief astronomical background

Tidal hydrodynamics deals with the forced oscillations caused by the astronomical tide-producing forces. The tides are the most predictable phenomenon in estuaries because of this astronomical forcing, which varies due to the well-known periodic motions resulting from the revolution of the Earth around the sun, the revolution of the moon-Earth system, and the rotation of the Earth on its axis (see *Tides*, and also Parker, 2007). These periodic motions determine that in any physical data set from an estuary (e.g., water level, currents, salinity, and water temperature), the tidal energy will be found at numerous very precisely defined tidal frequencies, whose amplitudes and phases can easily be calculated by harmonically analyzing that data set. There are dozens of these so-called tidal constituents, each constituent representing some aspect of the relative astronomical motions of the Earth, moon, and sun. These constituents tend to fall into bands. In the semidiurnal band, constituents produce two high waters (and two low waters) per day. In the diurnal band, constituents produce one high water (and one low water) per day. There are also other less important bands, including higher harmonics caused by shallow-water nonlinear effects, which will be discussed later in this entry. Tidal hydrodynamics modifies the amplitude and phases of these tidal constituents.

For the purposes of this entry, only the five most important tidal constituents will be mentioned until the shallow-water constituents are discussed (for more constituents see Parker, 2007, Table A, which includes 149 tidal constituents). These five larger constituents are named M_2, S_2, N_2, K_1, and O_1, the first three being semidiurnal and the last two diurnal. M_2, the main lunar semidiurnal constituent, represents the Earth turning under a slowly revolving moon with 2 cycles (and two high waters) in each 24.8412-h lunar day. It thus has a period of 12.4206 h and a frequency (1/tidal period) of 1.9323 cycles per (solar) day. S_2, the main solar semidiurnal constituent, represents the Earth turning under the sun with 2 cycles (and two high waters) in each 24.00-h solar day. It thus has a period of 12.00 h and a frequency of 2.00 cycles per (solar) day. When the moon and sun are in alignment at new and full moons, their tidal forces work together to produce larger tide ranges (called spring tides). When the moon and sun are out of alignment, at first and third quarters, their tidal forces work against each other to produce smaller ranges (called neap tides). N_2, with a period of 12.6583 h, represents the effect of the elliptical

Earth–moon orbit, in which the distance between the moon and Earth varies over a 27.5546-day period, from perigee (the moon closest to the Earth, and so a stronger tidal force) to apogee (the moon farthest from the Earth, and so a weaker tidal force) and back to perigee. This modulates the main lunar tidal force (M_2), and that modulation is represented by N_2 combined M_2. The diurnal lunar tidal forces resulting from lunar declination are represented by two diurnal tidal constituents, O_1 and K_1, with periods of 25.8193 and 23.9345 h (and frequencies of 0.9295 cpd and 1.0027 cpd). The minimum combined effect of these two constituents occurs every 13.66 days, at the times when the moon is over the equator. Their maximum combined effect occurs at maximum lunar declination, when the moon is either farthest north of the equator or farthest south of the equator. Because of these well-known astronomical frequencies, some form of tide prediction has existed since even the earliest civilizations. For a history of tide prediction, see Parker (2012).

Hydrodynamic effects on tide ranges and tidal current speeds in estuaries

When the very long tide wave generated in the deep ocean reaches the shallower water of the continental shelf and the even shallower water of estuaries, it is slowed up, amplified, modulated, and distorted by a number of hydrodynamic mechanisms. To understand what happens to a tide wave in an estuary, it is helpful to first look at two opposite extremes, the tide wave in a long river and the tide wave in a closed-off bay, both basins being deep enough for a minimal damping effect from bottom friction.

Progressive tide waves

The tide wave propagates up the river as a progressive wave (see Figure 1, top panel), which means that the crest of the wave (high water) moves progressively up the river, as does the trough of the wave (low water). In such a progressive tide wave, the maximum flood current (namely, when the current is flowing the fastest up the river) occurs at the same time as high water, and the maximum ebb current (namely, when the current is flowing the fastest down the river) occurs at the same time as low water. Slack water (when the current speed is zero) occurs exactly halfway between high water and low water. If the river is of constant width, the amplitude (tide range) of this progressive tide wave will not change as it moves up the river. However, if the width decreases as going upriver, then the amplitude (tide range) will increase, because the same amount of water is being forced through a smaller basin cross section. If the depth of the river decreases, there is a similar though less dramatic amplifying effect (which is generally outweighed by the increased energy loss from bottom friction due to the shallower depths).

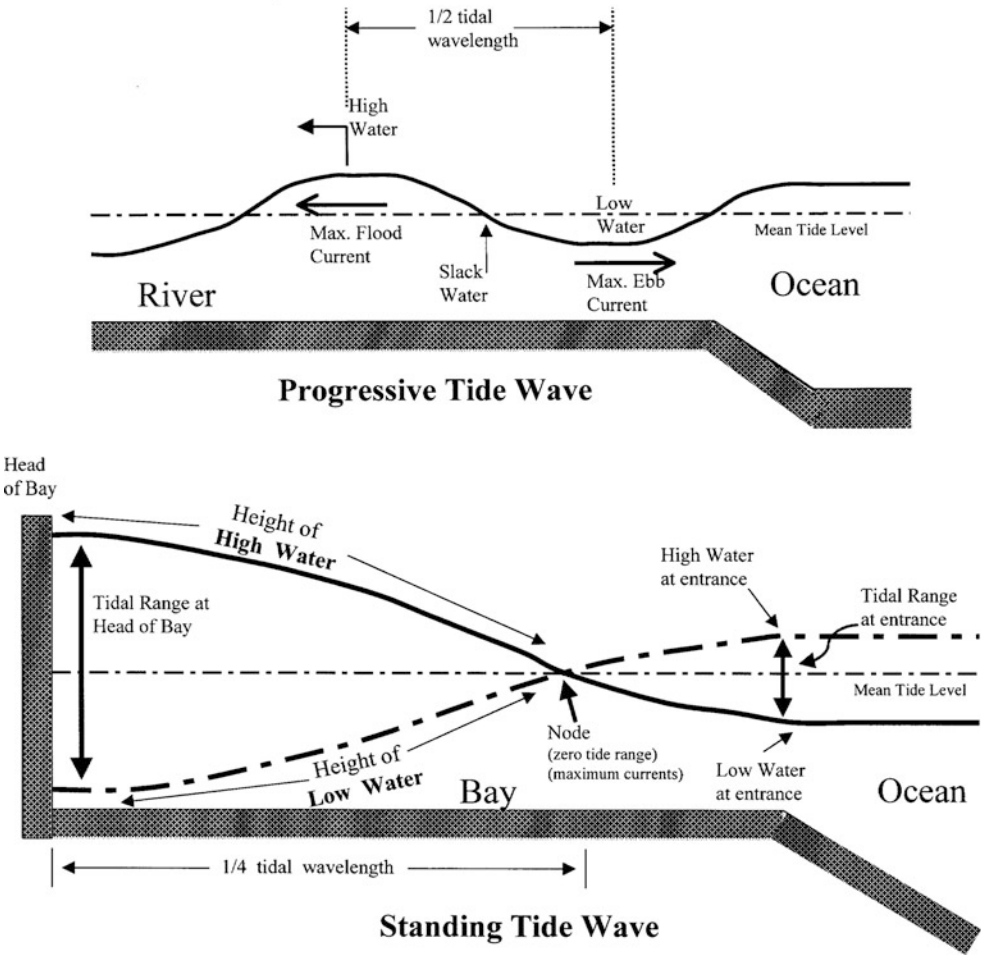
Standing tide waves

In a closed-off bay, the tide wave progressing up the bay will be reflected at the closed end and will travel back down the bay. This reflected wave is not observable by someone on the shore because it is superimposed on the incoming tide wave propagating up the bay, and it is the combination of the two waves that is observed. The resulting combined wave is called a standing wave, because the high and low waters do not progress up the bay (see Figure 1, bottom panel). The water surface simply moves up and down everywhere at the same time, with the largest tide range at the head of the bay. With a standing tide wave, the tide range increases as one moves from the ocean entrance toward the closed end of the bay, assuming the length of the bay is less than or equal to one fourth of a tidal wavelength from the head of the bay. If the bay is longer than that, there will be a location with minimum tide range (at one fourth of a tidal wavelength from the head of the bay), so from the entrance moving up the bay the tide range would first decrease and then increase. This location is explained by looking at the incident and reflected progressive waves. In a progressive wave, high water comes one half a wavelength before low water, so if the high water of a progressive wave travels a distance equal to one fourth of a tidal wavelength up the bay to the head, where it is reflected, and then travels one fourth of a wavelength back down the bay, it will have gone one half a wavelength and so coincide with low water of the incoming progressive wave, and the two will roughly cancel each other out at that location, producing the minimum tide range (if there was no friction and the incident and reflected wave were equal in amplitude, the minimum range would be zero and it would be called a node).

For a standing wave, high waters occur at the same time everywhere on one side of the minimum (node), which is the same time that low waters occur on the other side. The strongest tidal currents occur when water level is near mean tide level, halfway between the times of low water and high water. At the times of high water and low water, there is no current flow (slack water). The water flows into the bay, stopping the inward flow at high water; reverses direction; and flows out of the bay until low water, at which time it reverses again and starts flowing into the bay again.

Amplification of tides in an estuary

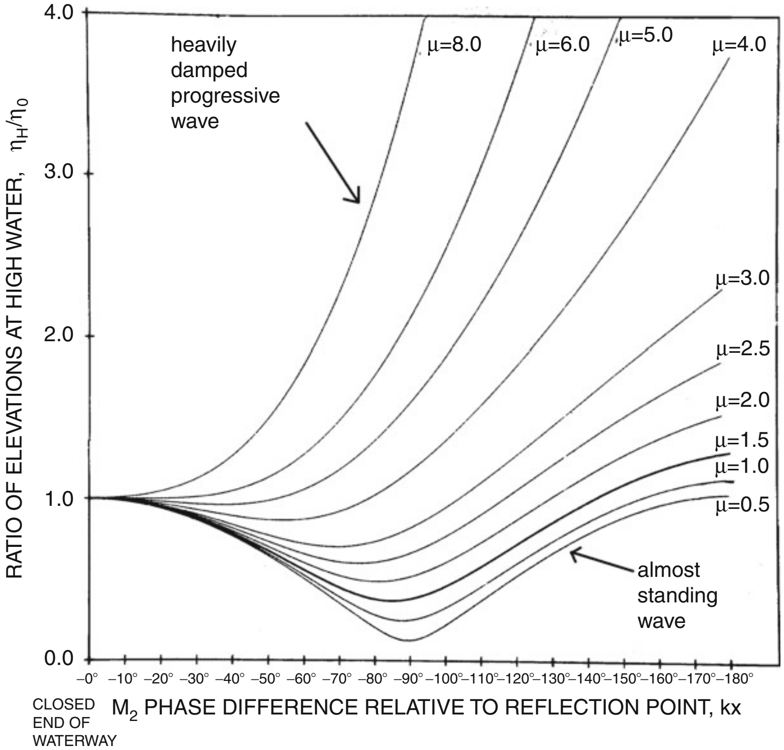
The tidal wavelength is determined by the depth of the estuary. If friction is ignored, the formula for the tidal wavelength, $\boldsymbol{\lambda}$, is $\lambda = T(gD)^{1/2}$, where D is the depth of the water, T is the tidal period, and g is the acceleration due to gravity. The shallower the bay, the shorter is the wavelength. When the length of the bay equals one fourth of a tidal wavelength, then the bay's natural period of oscillation will be the same as the tidal period. One finds the largest tide ranges in bays that are exactly one fourth of a tidal wavelength long, due to what is called resonance.



Tidal Hydrodynamics, Figure 1 (*Top panel*) An idealized frictionless tide wave propagating up a river as a progressive wave. High water occurs later as one moves upstream. (*Bottom panel*) An idealized frictionless tide in a bay as a standing wave (the water level is shown for two opposite extremes, high water and low water). High water occurs at the same time everywhere on one side of the node (the point of zero range).

When the water in the bay is forced to move up and down by the tide at the entrance, it will freely oscillate (slosh up and down and back and forth) with a natural period that depends directly on the bay's length and inversely on the square root of its depth, namely, $T = \lambda/(gD)^{1/2}$. If the basin has the right combination of length and depth so that the natural period is exactly the same as the tidal period, then the oscillation inside the bay will be synchronized with the oscillation at the entrance due to the ocean tide. In other words, the next ocean tide will be raising the water level in the bay at the same time that it would already be rising due to its natural oscillation (stimulated by the previous ocean tide wave), so that both are working together, thus making the tide range inside higher. (In the real world, friction keeps these resonating oscillations from being infinitely large, by taking away some of the energy.)

In the above discussion, bottom friction was considered minimal because of deep water, but bottom friction greatly affects all hydrodynamics and is especially important in shallow waters. Because of bottom friction, the tide wave in real estuaries actually falls in between the two extremes of the pure progressive wave and pure standing wave described above. This is because friction reduces the amplitude of the tide wave as it travels. Thus, the reflected wave will always be smaller than the incoming wave, especially near the bay entrance (since the reflected wave has traveled longer to get all the way back to the entrance), and the combination of these two frictionally damped progressive waves will not be a pure standing wave. There will be no point of zero tidal range (no node), but only an area of minimum tidal range (a quasinode). There will be some progression of high waters and low waters up the bay, but not as quickly as a pure progressive wave. This progression will be faster near the entrance and slowest near the head of the bay. Maximum flood or ebb currents will not occur exactly halfway between high water and low water. A basin one fourth of a wavelength long will still produce the largest possible tidal range at



Tidal Hydrodynamics, Figure 2 The amplitude ratio (=tide range ratio) plotted versus distance from the closed end of the waterway (extreme *left* of plot) in terms of wavenumber phase, kx, for several values of the frictional damping coefficient, μ. $kx = -90^o$ represents one fourth of a tide wavelength from the closed end of the bay.

the head of the bay, but friction will keep that tide range much smaller than it would be without friction. (Mathematical formulas representing the effects described above are derived in Parker, 2007).

Figure 2 illustrates how friction affects the tide range along a waterway. It is based on a relatively simple analytical model of an exponentially damped tide wave that reflects off the closed end of the waterway (Parker, 2007). In this figure, the ratio of the tide range (at any location) to the tide range at the closed end is plotted for various values of the damping coefficient μ. This is a convenient way to put many cases on the same plot, because every case can be plotted relative to the closed end. (Such things as amplification are easier to visualize if values are plotted relative to the entrance, but then each case would have to be plotted separately.) In Figure 2, if one imagines the entrance at, for example, one fourth of a tide wavelength from the closed end of the estuary (indicated by $kx = -90^o$ in the figure), then one can see that for $\mu = 0.5$ (which represents very little frictional damping in a deep waterway) the tide range increases significantly as one moves up the estuary toward the closed end (in the figure, moving to the left from $kx = -90^o$). At the other extreme, for $\mu = 8.0$ (great frictional damping in a very shallow estuary) the tide range decreases significantly as one moves up the estuary. For the case of very little frictional damping ($\mu = 0.5$), one sees an almost standing wave with a nearly zero tide range at one fourth of a tide wavelength ($kx = -90^o$), that location being the quasinode. For the case of very large frictional damping ($\mu = 8.0$), one sees an almost pure damped progressive wave that decreases in tide range as it moves up the waterway. (For this last case, it could also represent an open-ended river that is not very shallow, but for which there is no reflected wave because it is not closed off.)

In general, one sees very large amplification for small μ for waterway lengths that are equal to or less than one fourth of a tide wavelength ($kx = -90°$). For $\mu = 0$ (no friction), there is infinite amplification at exactly one fourth of a wavelength (namely, resonance). That never happens in the real world (because of friction), but very large tide ranges do occur in bays with lengths that are near a quarter tidal wavelength. For waterways that are shorter or longer than one fourth of a wavelength, the amplification is not as great (for the small μ case; for the large μ case there will never be amplification because of the strong frictional damping). From this figure, one can see that for a deep waterway (small μ) that is one half a tidal wavelength long ($kx = -180^o$), the tide range decreases until one reaches one fourth a wavelength and then increases until at the closed end of the waterway the tide range is almost the same as it was at the entrance.

The largest tides and fastest tidal currents

The largest tide ranges are found in bays that are close to one fourth of a tidal wavelength long. Tide ranges reach 15 m (50 ft) in Minas Basin in the Bay of Fundy and in Ungava Bay (also in Canada). Tidal ranges greater than 12 m (40 ft) occur at the northern end of Cook Inlet near Anchorage in Alaska, in Bristol Bay in the United Kingdom, in the Magellan Strait in Chile, in the Gulf of Cambay in India, and along the Gulf of St. Malo portion of the French coast bordering the English Channel. In some bays, the very high tide range at the head of the bay is due to a combination of both a narrowing width and a near resonant situation (due to the right length and depth). The highest tide ranges may involve several amplifications, the bay being perhaps connected to a gulf which is perhaps connected to a wide continental shelf, with amplifications of the tide wave occurring in each basin. This is the case with the Bay of Fundy tides, the tide wave being already amplified by the continental shelf and the Gulf of Maine prior to entering the Bay of Fundy. Huge tide ranges are not restricted to bays. If the continental shelf is the right combination of depth and width, a near resonant situation can also result. This is the reason for the 12-m (40-ft) tidal ranges along the coast of southern Argentina. The continental shelf there is over 965 km (600 mi) wide and includes the Falkland Islands near the edge of the shelf, where the tide range only reaches 2.0 m (6.5 ft). The distance from the Argentinean coast to the edge of the shelf is fairly close to one fourth of a tidal wavelength for that depth of water. Essentially, that wide shelf has a natural period of oscillation that is fairly close to the tidal period.

The largest tidal currents in estuaries tend to be near the entrances. (Maximum tidal current speeds are zero at the head of the bay, since there is no place for the water to flow). As one moves down an estuary toward the ocean, the maximum flood and maximum ebb tidal current speeds increase, with the greatest speeds occurring at the entrance, or, if the estuary is long enough, at the area of smallest tide range (the nodal area). However, if the width of the estuary decreases at any point, the current speeds will be increased in that narrow region. (It must flow faster, since the same volume of water is being forced to flow through a smaller cross section). This can be especially dramatic if there is a sudden decrease in width and depth. The largest tidal currents are found in narrow straits in which the tides at either end have different ranges or times of high water. Tidal current speeds greater than 7.7 m/s (15 knots) occur in Seymour Narrows, between Vancouver Island and the mainland of British Columbia, Canada. Tidal currents of 5.1 m/s (10 knots) are found in South Indian Pass in Southeast Alaska and in Kanmon Strait, Japan. Fast tidal currents in narrow straits that suddenly widen can also produce a tidal whirlpool, a violently rotating funnel-shaped hole of water, which is explained later in this entry.

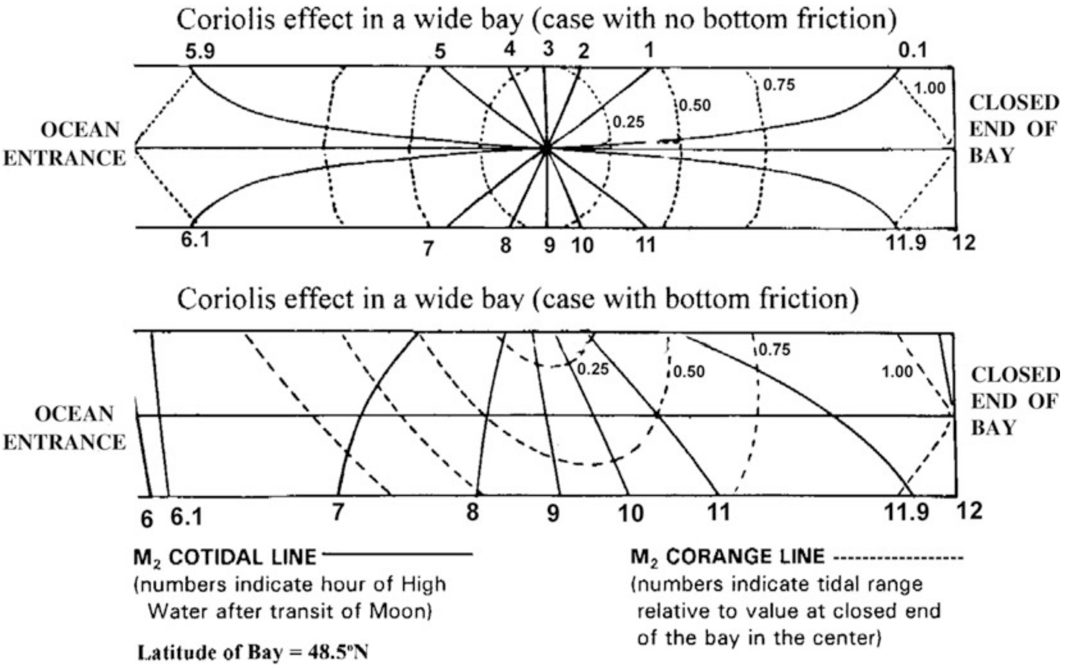
Coriolis effects on tide ranges in wide estuaries

If an estuary is wide enough, one also sees larger tide ranges on the right side of the bay (looking up the bay) due to the Coriolis force. The Coriolis force is a fictitious force due to our observing motion from the rotating reference frame of the Earth (see Parker, 1998). It acts perpendicular to the flow of the water, thus pushing water currents to the right in the Northern Hemisphere (and to the left in the Southern Hemisphere) as they flow. For a pure progressive tide wave (no friction), at high water the tidal current flows up the bay, so the tidal height will be greater on the right-hand shore than on the left-hand shore. At low water, the tidal current flows down the bay, so the tidal height will be lower on the right-hand shore than on the left-hand shore. The result is that the tide range (high water minus low water) will be greatest on the right-hand shore (looking up the bay). For a pure standing wave (no friction), the pattern of high water caused by the Coriolis force is more complicated, as is shown in the upper half of Figure 3. This figure shows lines of constant tide range (corange lines), as well as lines of locations with the same time of high water (cotidal lines), in an idealized rectangular basin for the case where the effect of bottom friction is ignored. A single point of zero tidal range (a node) occurs in the center of the bay one fourth of a tidal wavelength from the head of the bay.

This figure comes close to representing the corange and cotidal lines in a bay that is very deep. A more typical case, including the damping effect of bottom friction, is shown in the bottom half of Figure 3. In this case the node has moved to the left (when looking up the bay) and becomes a virtual node since it is on land. (See Parker, 2007, for derived mathematical formulas describing this effect.) One can see some similarity between the pattern of corange lines in the lower half of Figure 3 and the M_2 corange lines in Figure 4 for the Strait of Juan de Fuca–Strait of Georgia (although the latter has many geographic variations not included in the simple regular basin of Figure 3). In Figure 4 the pattern of a quasinode is also seen to the southwest of Victoria.

Hydrodynamic effects on diurnal versus semidiurnal tides in estuaries

In many estuaries it is common to have semidiurnal tides, namely, two high waters (and two low waters) a day, with the heights of the two high waters (and two low waters) being approximately the same size. In some estuaries and seas, there can be diurnal tides, namely, only one high water (and one low water) per day. Most common, however, is to have mixed tides, namely, two high waters (and two low waters) a day but where there is considerable difference between the heights of two consecutive high waters (and/or between the heights of the two consecutive low waters), that difference being due to a strong diurnal signal. It is tidal hydrodynamics that determines which of these three types of tide one will see at particular



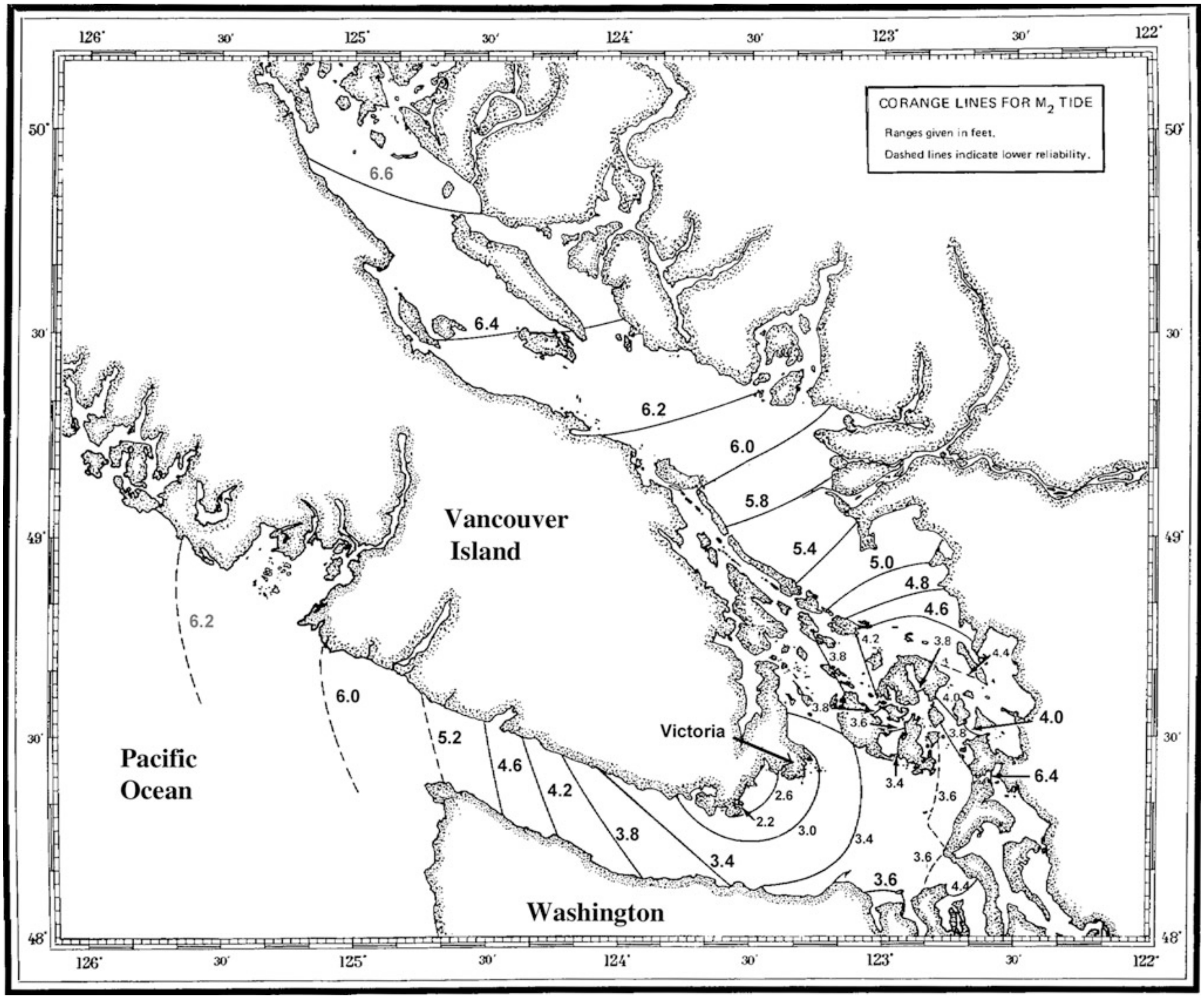
Tidal Hydrodynamics, Figure 3 The effect of Coriolis force on the M_2 tide range (*corange lines*) and the time of high water (*cotidal lines*) for an idealized rectangular bay. The *top panel* shows the case with no *bottom* friction; the *bottom panel* includes the effect of *bottom* friction.

locations in an estuary. This classification of tides is not always as descriptively precise as one might wish, because the relative strength of the diurnal signal to semidiurnal signal varies throughout the month, with the diurnal signal being strongest during maximum lunar declination (i.e., when the moon is farthest north or south of the equator). Thus, a so-called mixed tide might have two high waters a day when the moon is over the equator, but only one high water a day when the moon is farthest north or south of the equator (and thus being diurnal at those times).

The dimensions of a basin (and the dimensions of the waterways leading to the basin) determine the size of the diurnal tidal signal (primarily K_1 and O_1) compared with the usually dominant semidiurnal tidal signal (primarily M_2, S_2, and N_2). A particular bay could have a natural period of oscillation that is closer to the diurnal tidal period (approximately 24.84 h) than to the semidiurnal period, thus amplifying the diurnal forcing at the entrance to the bay more than the semidiurnal signal. Depending on the size of the diurnal signal at the entrance, the result could be a mixed tide or a diurnal tide. At such locations (e.g., parts of the Gulf of Mexico), the tide will be diurnal near times of maximum lunar declination, but will be mixed near times when the moon is over the equator.

As mentioned earlier, the wavelength, **λ**, of a tide wave in a bay depends on the depth of the water, D, and on the tidal period, T, according to $\lambda = T(gD)^{1/2}$ (if frictional effects are ignored). The longer the tidal period, the longer is the tidal wavelength. A diurnal tidal component has a wavelength twice as long as a semidiurnal tidal component, since its period is twice as long. When a waterway is shallow enough and long enough so that more than one fourth of a semidiurnal (e.g., M_2) wavelength fits in the waterway, there will be a nodal area with a very small semidiurnal (M_2) tidal range. This will be an area where the diurnal tide could dominate, since the diurnal tide would still be large at the semidiurnal nodal area (the diurnal node being twice as far from the head of the bay). Thus, near the head of the waterway the tide could be semidiurnal, but near the semidiurnal nodal area, the tide could be mixed or even diurnal. This is the case near Victoria, British Columbia, at the southeastern end of Vancouver Island (see Figure 5). At that location along the Strait of Georgia–Strait of Juan de Fuca waterway, the M_2 and S_2 tidal constituents decrease to a minimum, but the K_1 and O_1 constituents do not, and so the tide becomes mixed mainly diurnal, while at the northern end of the Strait of Georgia, the tide is mixed, mainly semidiurnal.

Whether due to a basin size conducive to amplifying the diurnal signal or due to the existence of a semidiurnal nodal area (leaving the diurnal signal as the dominant one), there are numerous areas around the world with strong diurnal tides – places like Norton Sound in Alaska near the Bering Strait and various (but not all) locations in the Philippines, New Guinea, and the islands of Indonesia. In southern China, at Beihai, and at Do Son, Vietnam, the diurnal signal is very dominant, with diurnal tidal ranges that reach 4.6 m (15 ft) and 3.0 m



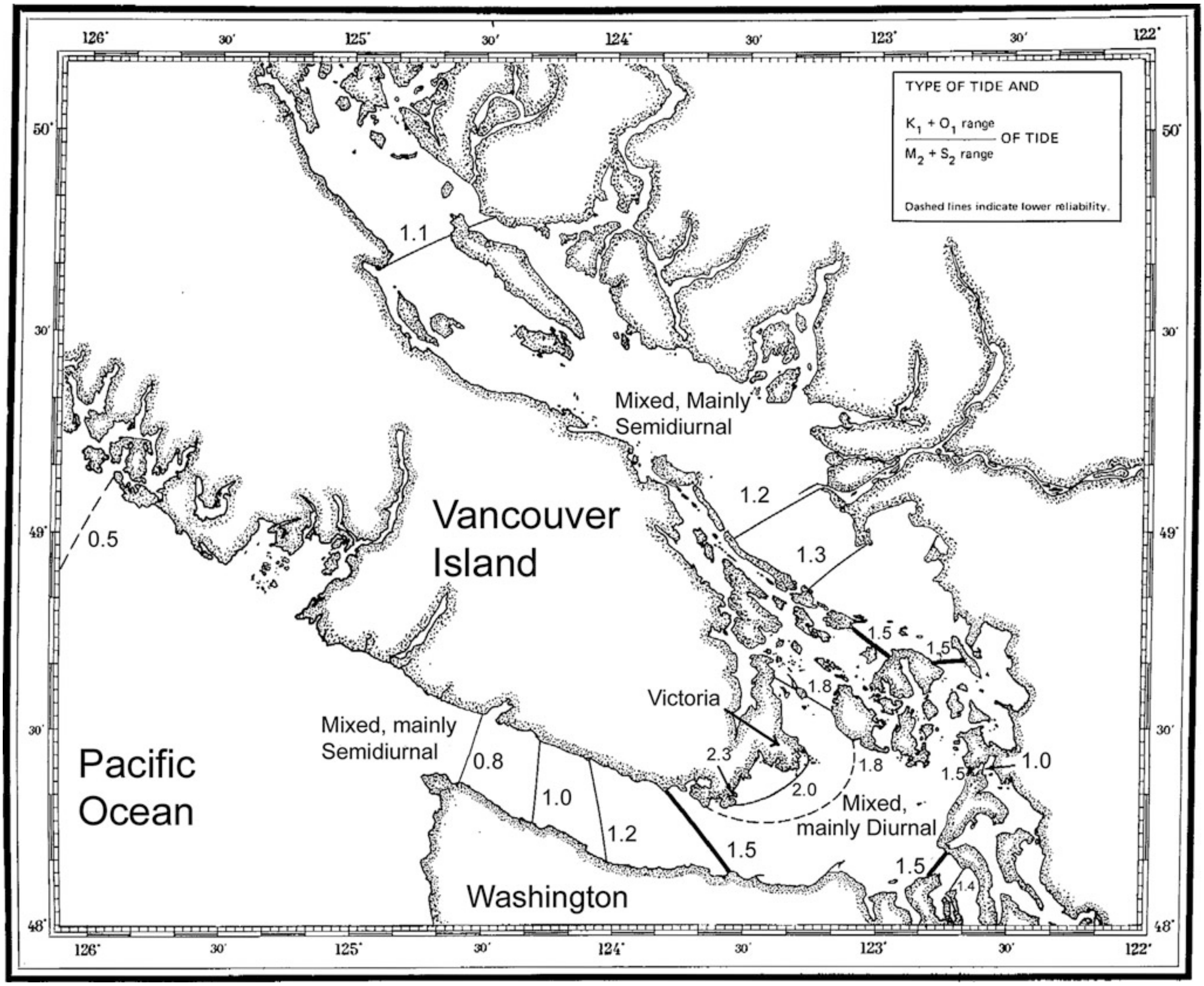
Tidal Hydrodynamics, Figure 4 A corange chart showing the geographic variation of the M_2 tide range for the Strait of Juan de Fuca–Strait of Georgia. A minimum tidal M_2 range occurs just to the southwest of Victoria, British Columbia (1 ft = 0.305 m).

(10 ft), respectively (near times of maximum southern declination of the moon). In these locations, the tide remains diurnal even when the moon is over the equator.

Nonlinear effects of shallow water on tides: overtides and compound tides

Shallow water also has other effects on the tide. It can, for example, distort the shape of the tide wave, that is, make it very asymmetric, so that its rise and fall (and its flood and ebb) are no longer equal (see the second curve in Figure 6). The tide can then no longer be described by a simple sine wave (such as the first curve in Figure 6). In some cases, such distortion leads to double high waters or double low waters (see the third curve in Figure 6). The extreme case of distortion is a tidal bore (the fourth curve in Figure 6), when the tide wave becomes so steep that it is essentially a continuously breaking wave moving up a river as a turbulent wall of water.

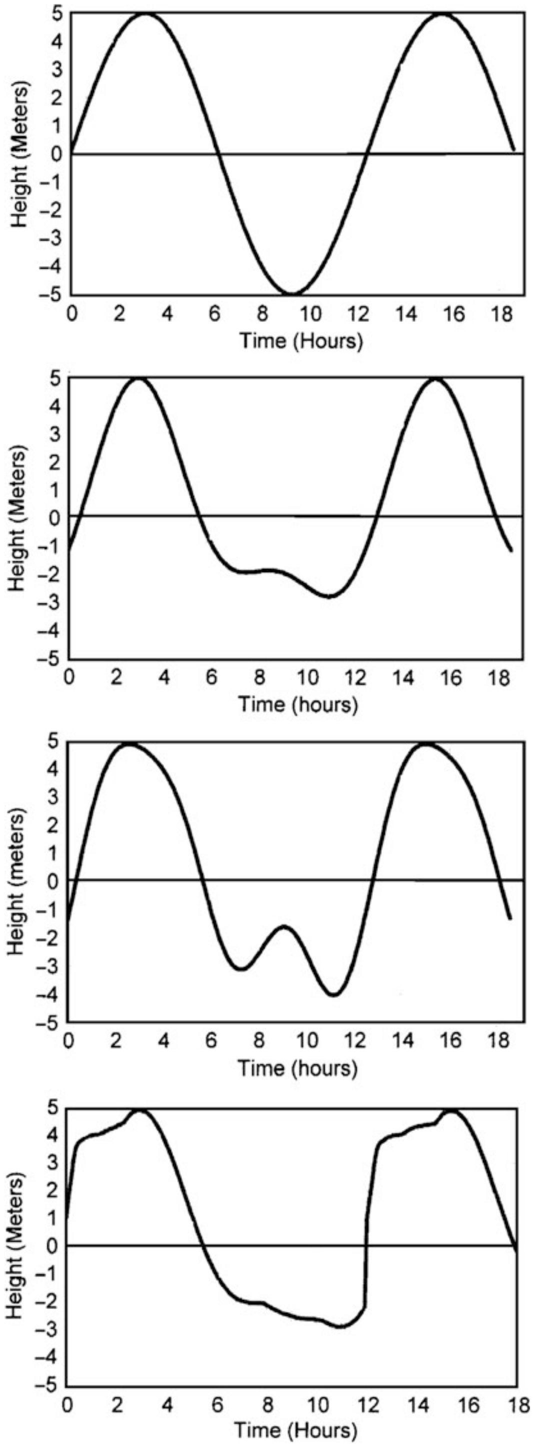
Shallow water distorts the tide through several mechanisms that are nonlinear – that is, in the equations of motion (based on conservation of mass and momentum, see Parker, 2007), each mechanism can be tied to a specific term in which key parameters (such as water elevation or velocity) multiply each other, which leads to energy transfer. (Linear terms contain only one key parameter and thus their separate effects simply add, with no interaction.) Such nonlinearities can only be handled adequately in numerical hydrodynamic models (based on the nonlinear equations of motion), rather than by simple formulas from analytical models, such as those that we used to described many of the linear tidal effects.



Tidal Hydrodynamics, Figure 5 Chart illustrating the geographic variation in the $(K_1 + O_1)/(M_2 + S_2)$ tidal constituent amplitude ratio, as well as the *type of tide* classification for each region in the Strait of Juan de Fuca–Strait of Georgia.

However, the effects of these nonlinear terms can be explained physically here without resorting to the math of these nonlinear models. (For mathematical treatments of the nonlinear terms, see Parker, 2007).

The speed, C, at which a long tide wave travels depends on the depth of the water, D, approximately (ignoring friction) as the formula $C = (gD)^{1/2}$. When the depth of the water is much greater than the tidal range, the speed of the crest of a tide wave and the speed of the trough are virtually the same, since the tide wave itself has only a very small effect on the total water depth. However, in the shallow water where the depth is not much greater than the tide range, the total water depth under the crest is significantly larger than the total water depth under the trough. In this case, the crest of the wave (high water) travels faster than the trough of the wave (low water). If the tide wave travels far enough, the crest begins to catch up with the trough ahead of it (which is falling behind the crest ahead of it). Thus, high water arrives sooner than it would in deeper water, and there is a faster rise to high water and a slower fall to low water. The shape of the tide curve could perhaps look like that shown in the second curve in Figure 6. In terms of harmonic constituents, this distortion transfers energy from M_2 into the second harmonic, a constituent called M_4, with half the period of M_2. Combining an M_2 tide curve and an M_4 tide curve, one can produce the distorted tide curves shown in Figure 6, with the M_4/M_2 ratio increasing as one goes from top curve to bottom curve in those figures. The third curve in Figure 6 shows a double low water, but with a different phase relationship between M_2 and M_4, one could obtain a double high water.



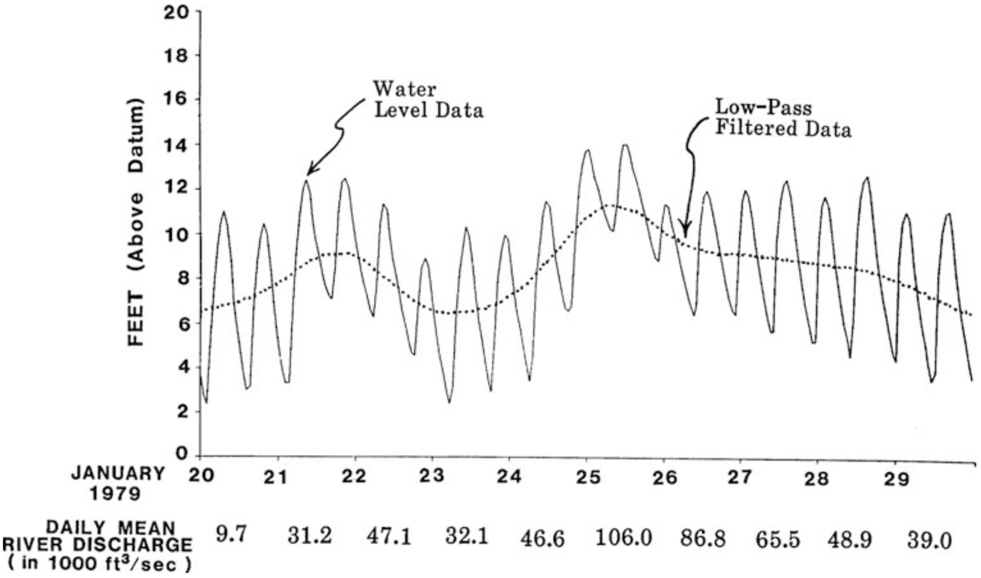
Tidal Hydrodynamics, Figure 6 Typical tide curves (over one and a half tidal cycles) for an area with (*top panel*) no shallow-water effect and for three areas with increasing degrees of distortion caused by the shallow water. The *second panel* shows moderate distortion of the sinusoidal curve seen in the first panel. The *third panel* shows a double low water. The *fourth panel* shows the almost instantaneous rise in water level due to the passage of a tidal bore.

Another shallow-water distorting mechanism is caused by bottom friction, which can have both asymmetric and symmetric effects. The asymmetric effect (similar to that just discussed and represented in Figure 6) is produced because friction has a greater effect in shallow water than in deep water (there being less water to have to slow down), and so it slows down the trough more than the crest, contributing to the distortion of the tide wave and the generation of M_4. The symmetric effect is produced because frictional energy loss is proportional to the square of the current speed. This means that there will be much more energy loss during times of maximum flood and maximum ebb than near times of slack water (or minimum flow). This results in the generation of another higher harmonic, M_6, with a period of one third that of M_2. This effect, combined with the asymmetric effect, can lead to double high or low waters (such as the third curve in Figure 6).

Higher harmonic tidal constituents like M_4 and M_6 are referred to as overtides (a term analogous to the term overtones in acoustics). M_4 is the first overtide (and the second harmonic) of M_2. M_6 is the second overtide (and the third harmonic) of M_2. M_8 is the third overtide (and the fourth harmonic) of M_2. Whereas M_4 and M_6 are generated by first-order nonlinear processes, M_8 is generated by second-order nonlinear processes (i.e., it is generated by nonlinear effects on M_6).

Friction dissipates energy from the entire tide wave and slowly wears the entire tide wave down. However, if, as the tide wave propagates up the river, the river's width is decreasing significantly, this can keep the amplitude of the wave high in spite of the friction. Thus, the tide wave can continue to travel up a narrowing river, getting more and more distorted in shape. A further distortion can be caused by the river flow interacting with the tide (see below). In the extreme case, the distortion from all these effects can lead to the creation of a tidal bore, when the tide wave becomes so steep that it is essentially a continuously breaking wave, moving up a river as a turbulent wall of water (see fourth curve in Figure 6). (The earliest known tide table was printed in 1056 AD for the tidal bore on the Qiantang River in China; see Parker, 2012.)

New tidal constituents called compound tidal constituents can be produced through the nonlinear interaction of two astronomical tidal constituents. For example, the above symmetric quadratic friction effect causes the interaction of two tidal constituents, such as M_2 and N_2. M_2 and N_2 go in and out of phase over a 27.6-day cycle (perigee to apogee to perigee). In this case the greatest energy loss occurs when M_2 and N_2 are in phase and producing the strongest tidal currents, and the lowest energy loss occurs 13.8 days later when M_2 and N_2 are out of phase and producing the weakest tidal currents. Because energy loss is proportional to the square of the current speed, the increased energy loss when M_2 and N_2 are in phase is greater than the decreased energy loss when they are out of phase, and the result is that each constituent will be smaller than if it existed without the other present. The



Tidal Hydrodynamics, Figure 7 Water level data from the tide gauge at Trenton, NJ, during a high river discharge period (January 20–29, 1979). The tide range is reduced when the river discharge is high, and the tide curve is distorted (with a faster rise to high water) (1 ft = 0.305 m).

reduction in N_2 (due to M_2) will be greater than the reduction in M_2 (due to N_2), because M_2 is much greater than N_2. However, M_2 will be reduced by the combined interactions of all the other tidal constituents. There is a 27.6-day modulation of this energy loss from M_2 and N_2, and this produces two new compound tidal constituents called $2MN_2$ and $2NM_2$. (Similarly, the above asymmetric mechanisms also cause interactions between constituents, producing higher frequency constituents such as MN_4 from M_2 and N_2.) More compound tides and overtides and their origin are explained by Parker (2007).

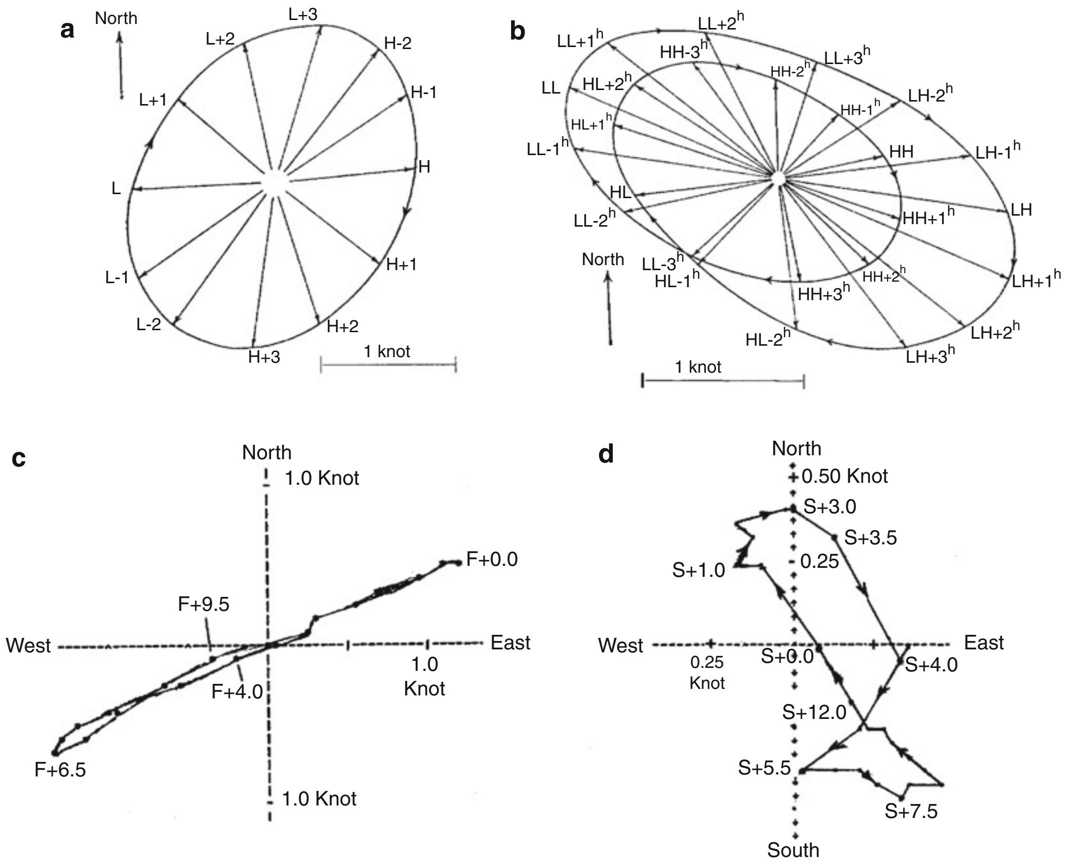
In shallow waterways with large tidal ranges, the nonlinear effects can produce dozens of compound tides and overtides of significant enough size that they must be included in harmonically based tide predictions. For example, for tide predictions at Anchorage, Alaska, at the northern end of Cook Inlet, 114 tidal constituents (most of them shallow-water constituents) must be used to predict the 30-ft tides there (as compared with typically 26 or less constituents in other waterways).

Nonlinear tidal interaction with river flow

In a tidal river, water flow is due to both the tidal current and the river current itself (i.e., the freshwater flowing downhill). The result of the combined tidal current and river current is a faster and longer-lasting ebb current phase and a slower and shorter flood current phase. Far enough up a river, where the river flow is faster than the strongest tidal current, the flow of water will always be downstream. In this case, the speed of flow will oscillate, flowing the fastest downstream at the time when maximum ebb occurs farther down the river and flowing the slowest downstream at the time when maximum flood occurs farther down the river. This is a simple linear addition of the river current to the tidal current.

However, because of the shallow water, the river flow also interacts with the tide nonlinearly and distorts it, mainly due to the effect of bottom friction. Energy loss due to friction is proportional to the square of the total current speed. During ebb, the tidal current is in the same direction as the river current, and the result is a larger combined ebb current, with increased energy loss. During flood, the tidal current is in the opposite direction as the river current and the result is a smaller combined current, with reduced energy loss. This has an asymmetric effect that distorts the tide (causing a faster rise to high water, delaying the time of low water, and increasing the size of M_4). It also wears down the entire wave further because the increased energy loss during ebb is larger than the decreased energy loss during flood. In Figure 7, one can see the tide range shrink when the river discharge increases. The curves also become more asymmetric, rising more quickly to high water and falling more slowly to low water. A harmonic analysis of water level data during this period of high river runoff would give a smaller M_2 and larger M_4 than during times of small runoff. (See Parker, 2007, for a mathematical treatment of this effect and more examples.)

Another type of shallow-water effect causes interactions between the tide and low-frequency storm surges (generated by the wind) that have periods longer than tidal periods. In this case, when the water level is raised by an onshore wind, the water depth increases and changes the tidal dynamics, usually increasing the tide range. When an offshore wind lowers the water level, decreasing the



Tidal Hydrodynamics, Figure 8 Examples of three rotary tidal currents and one reversing tidal current (third example). The first two are off an ocean coast and are idealized. The last two are within an estuary and are from actual data. For the first two examples, each vector shows the speed and direction of flow for that hour of the tidal cycle, but for the last two examples only the tips of the vectors are shown.

water depth, the result is usually a decreased tidal range. (See *Tidal and Nontidal Oscillation*; see also Parker, 2007, for a mathematical treatment of this interaction.)

Tidal effects on mean sea level: another nonlinear effect

Tidal heights are referenced to some type of datum, typically the mean level of a particular key point on the tide curve. Datums such as mean lower low water (MLLW) or mean high water (MHW) or mean tide level (MTL) are usually calculated by averaging over 19 years of data (to eliminate the 18.6-year lunar nodal variations as well as meteorologically caused noise). Such datums may slowly change over the years due to a change in sea level (due to climate change) or due to slow vertical land movement (e.g., due to glacial rebound or sediment compaction), the latter looking like a sea level change because the water level gauge is mounted on the moving land. All the datums slowly move up and down with the long-term changes in relative sea level.

However, in shallow-water areas, there can also be a tidally induced change in "mean sea level" due to nonlinear tidal effects. In this case, what looks like a "mean" result comes from averaging an asymmetry within a tidal cycle caused by the nonlinear effects, rather than from a uniform shifting up or down of the water level on which the tide propagates. But this is an effect that is still included in the datum calculation. It is therefore an effect that should be considered when assessing long-term changes in sea level in a shallow-water area, since something as simple as dredging or shoaling can change the tidal hydrodynamics of a waterway and then also affect the value of mean sea level (and thus all the tidal datums). As one example, in the Delaware River and Bay, shallow-water nonlinear effects increase the mean sea level value near Philadelphia by an amount equivalent to 8 % of the M_2 amplitude at the entrance to the bay (Parker, 2007).

Tidal currents do not require datums, the zero current speed serving that purpose (if there is no mean current from a nontidal source). If there is a mean permanent current, such as due to a mean river flow or a mean wind drift, then the tidal current oscillates about that mean flow. Or, as seen in Figure 8 (fourth example), the entire tidal current ellipse can be shifted in the direction of the mean flow. But here again, the mean current can also be affected by the nonlinear tidal hydrodynamics causing an asymmetry within the tidal cycle. This can be due not only to shallow-water effects but even more importantly to nonlinear lateral inertia effects (that do not need shallow water). Such tidally induced residual current effects are discussed below, as one of the many phenomena that appear in tidal currents but not in the tide.

Special aspects of tidal currents (compared with the tide)

The tide is a scalar quantity, with a one-dimensional movement, that is, simply moving up or down. The tidal current, however, is a vector quantity, representing the horizontal water flow in two dimensions, and because it is a vector quantity, its movements are more complicated than the tide (but still very predictable because it has the same astronomical forcing).

There are situations where the tidal current flow can look approximately one dimensional, namely, when the estuary or waterway is very narrow. In this case, a plot of changing tidal current looks like a tide curve except the sine curve is above and below a zero speed line. This is called a reversing tidal current. In one tidal cycle, the current goes from its maximum positive value (maximum flood) through the zero current speed (slack water) to the maximum negative value (maximum ebb) and back again through slack water to maximum flood. The slacks are usually referred to as slack before flood (SBF) and slack before ebb (SBE). [Alternative names for the above include flood strength, ebb strength, slack flood begins (SFB), and slack ebb begins (SEB)].

Most tidal currents, however, are rotary tidal currents, that is, the direction of flow rotates 360° (around the compass) over one tidal cycle. This rotation is due to both the Earth's rotation and certain geographic conditions. Because the tidal current speed varies as this rotation takes place, the shape that the tip of the current vector traces out can be an ellipse of various widths or even almost a circle (in the open ocean) when the speed remains approximately the same throughout the cycle (Figure 8). Within an estuary that has a more complicated geography, for example, those with islands and various channels, the shape can even be more complicated (such as the shape shown in the fourth plot in Figure 8). Because of the elliptical shape, one can still use the terms flood current (for the tidal current flow into and up an estuary) and ebb current (for the tidal flow down and out of an estuary). However, there will not be a true slack water, because for that brief time, the flow will be cross-channel, so we use the terms minimum before flood (MBF) and minimum before ebb (MBE).

Dramatic spatial variation in tidal currents: vertically

The spatial variation of tidal currents, both horizontally (i.e., geographically) and vertically in the water column, is much more complex than the spatial variation of tides. The tide, being the one-dimensional movement of the water surface, does not have a variation with depth (although the vertical tidal movement of different water density layers does change considerably with depth). However, tidal currents vary considerably from the surface to the bottom. Bottom friction is a major cause of this variation. The tidal current is slowest near the bottom and faster nearer to (but usually not right at) the water surface. Usually the tidal current will turn (from flooding to ebbing, or vice versa) earlier near the bottom than near the water surface, so there will be times when the tidal current is going in the opposite direction at one depth than at another depth. One example of the variation in tidal current speed across a cross section of Chesapeake Bay is shown in Figure 9. In this figure, one sees the current flooding (i.e., flowing up the bay) in the deeper depths and ebbing (i.e., flowing down the bay) near the water's surface (except on the west side, where it is still flooding).

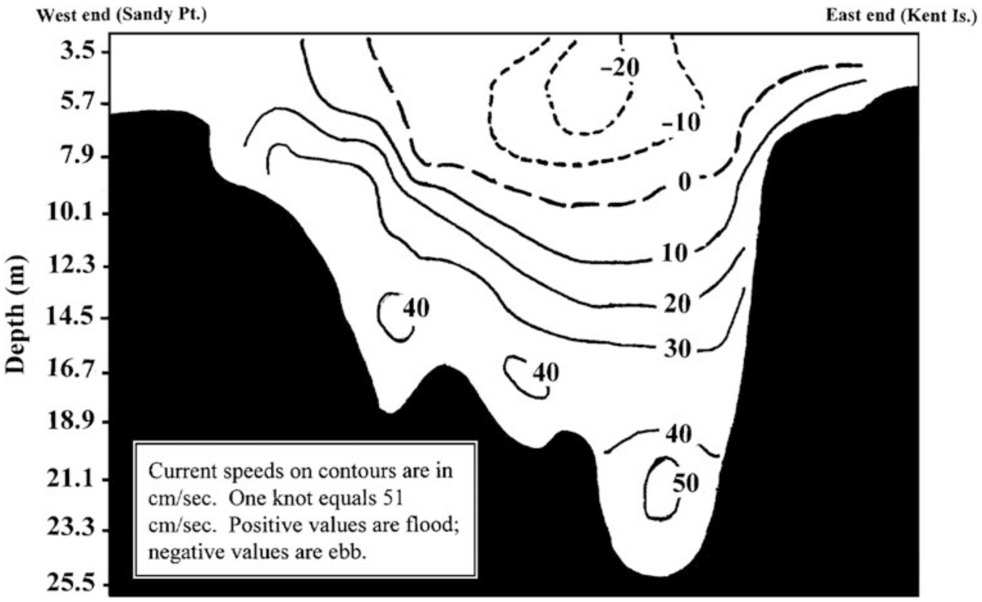
There are also other effects that can make the tidal current vary vertically. Tidal currents can be modified by baroclinic effects, that is, by density differences vertically along the water column, due to salinity differences (in estuaries that are not well mixed) or temperature differences (offshore in the coastal ocean), which allow the propagation of internal tide waves. Such baroclinic effects on tidal currents are often seen in the middle of a stratified water column, but not near the bottom or near the water surface, where frictionally caused mixing takes place.

Dramatic spatial variation in tidal currents: horizontally

Horizontally (i.e., geographically), the tide varies quite smoothly due to the hydrodynamic effects of such things as changing depths and widths, resonance, and Coriolis force. Such variations can often be reproduced or predicted with even simple analytical models. Although tidal currents are also affected by these same hydrodynamic effects and can change in similar ways, there are other hydrodynamic effects which can make tidal currents change dramatically in the horizontal direction, often over surprisingly short distances. One example is the tidal current in a navigation channel compared with the tidal current in the nearby shallows. The tidal current is much faster in the deeper channel than in the shallows, and the times of slacks and of maximum floods and ebbs can be quite different than those in the nearby shallows.

One does not require a dramatic change in depth to see differences in the tidal currents. Even in a wide bay, changes in bathymetry will affect the tidal currents. Not only will current speeds vary with horizontal distance (as mentioned above), but the bathymetry will steer the current. Also, currents within a channel or constricted portion of bathymetry will tend to be more reversing than currents in a more open and flat part of the bay, where a more rotary tidal current will be possible.

In Figure 9, where vertical variation in the tidal currents was seen, there was also horizontal variation along the width of the bay. Bottom friction is often the main reason for the horizontal variation in tidal current, due to the



Current speeds on contours are in cm/sec. One knot equals 51 cm/sec. Positive values are flood; negative values are ebb.

Tidal Hydrodynamics, Figure 9 Current flow at one moment in time through a cross section near the Chesapeake Bay Bridge. The current is flooding up the Bay in the deeper depths and ebbing down the Bay near the water's surface (except on the west side, where it is still flooding).

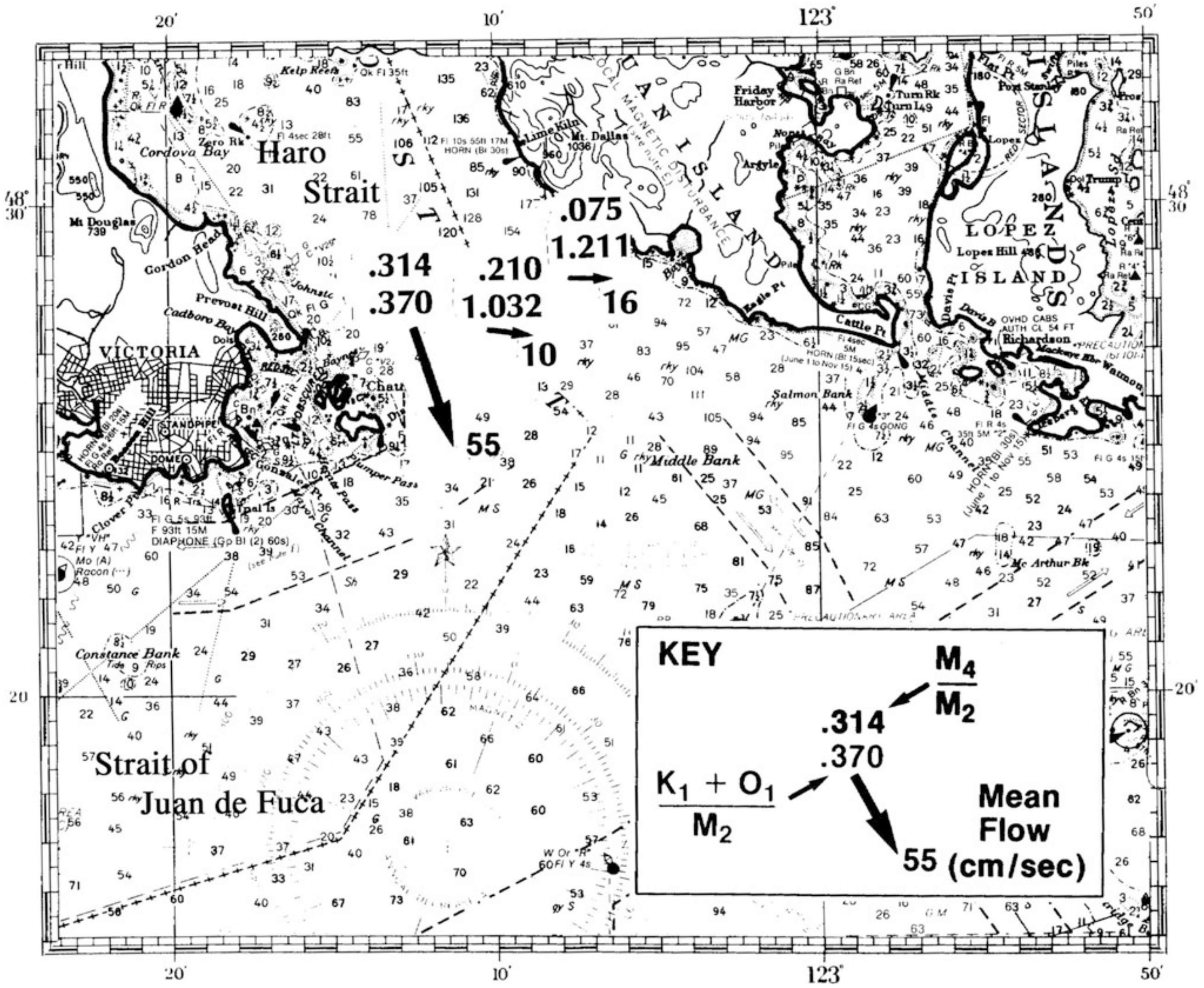
changing water depth as one moves horizontally. However, there are many other causes of horizontal variation in tidal currents, such as that discussed in the next section.

Nonlinear lateral inertial effects on tidal currents

One effect that can be even more dramatic than the frictional effect (due to changing depths) is found where there is a bending waterway, a channel bend, a point of land projecting into the waterway, or some other similar geographic variation in the shoreline. Such a feature can cause the formation of a large eddy during one or both phases of the tidal current. A point projecting into a waterway produces a large eddy on the side of the point sheltered from the tidal current. During the flood phase, there will be a large eddy on the backside of the point, and during the ebb phase, there will be a large eddy on the front side of the point. This is a lateral inertial effect. After the ebb phase ends, for example, the water on the backside of the point keeps moving roughly in the same flood direction, because that location is sheltered from the opposing flood currents by the point of land, and inertia keeps the sheltered water moving (Parker, 2007).

If one harmonically analyzes current data from a location within this eddy, one will obtain a consistent mean current, which is usually referred to as a tidally induced residual flow (and the process that causes it is often called tidal rectification). However, one will also see that the size of the tidal harmonic constants is very different than those for the waters not sheltered by the point of land, because the inherent asymmetry leads to energy being transferred to the second harmonics of the tidal constituents. Thus, the ratio of semidiurnal-to-diurnal constituents in the tidal current will be larger in sheltered locations than in unsheltered locations (and thus, the typically shown diurnal-to-semidiurnal constituent ratio will be smaller). Similarly the ratio of quarter-diurnal tidal current constituents to semidiurnal tidal current constituents will also be larger, leading to distorted tidal current curves. None of this affects the tide, and one will not see such dramatic variations in tide constituents across the waterway.

This inertial effect is one of many nonlinear effects which can modify tidal currents; however, this effect does not depend on the water depth being shallow, as most nonlinear effects do. When looking at the two-dimensional or three-dimensional momentum equations (Parker, 2007), it is the lateral advective/inertial terms that produce the tidally induced residual current and the transfer of energy to higher frequency tidal constituents. Figure 10 shows a good example of these inertial effects on tidal currents, for the region where the Strait of Juan de Fuca meets Haro Strait (between Canada and the United States). Both are deep waterways, but they meet at an angle, namely, the waterway bends significantly to the north near Victoria, BC. Thus, the flood current in the Strait of Juan de Fuca flowing past Victoria cannot make a sharp left turn because inertia keeps it moving eastward, allowing the southerly ebb current at the westernmost current station in Haro Strait to keep moving longer than at the current station on the eastern side of the waterway. Figure 10 shows harmonic analysis results from three current stations across the entrance to



Tidal Hydrodynamics, Figure 10 Variation in $(K_1 + O_1)/M_2$ and M_4/M_2 tidal current amplitude ratios, and in tidally induced residual currents, at three stations across the entrance to Haro Strait at a depth of 70 ft (21 m) below MLLW.

Haro Strait (entering from the Strait of Juan de Fuca). The $(K_1 + O_1)/M_2$ amplitude ratio in the tidal current varies dramatically, decreasing from 1.21 on the eastern side of the waterway to 0.37 on the sheltered western side (while for the tide the $(K_1 + O_1)/M_2$ ratio does not change much, only from 2.25 to 2.50). The M_4/M_2 ratio for the tidal current also varies significantly across the waterway, from 0.075 on the eastern side to 0.314 on the western side.

Tidal whirlpools

The most dramatic horizontal variation in a tidal current occurs in narrow straits (where the tidal current is very fast) that suddenly widens out, reducing the flow eventually but also producing a tidal whirlpool (also called a maelstrom). This is a violently rotating funnel-shaped hole of water that can suck ships underwater. To briefly explain how the tidal whirlpool is generated, we first note that when the water flows from the wide part of the strait into the narrow part, the current follows the shoreline and converges to flow more rapidly in the narrow part. But when the tidal current reverses and flows from the narrow part of the strait into the wide part, the rapidly flowing current cannot suddenly make the sharp left or right turn that would spread the flow over the whole width of the strait. The current's inertia keeps its flow going approximately straight down the middle. This allows water near the edges of the strait to continue moving in the same direction it had been flowing (also due to its inertia), until the bending shoreline forces that edge flow to bend to the middle and meet the flow down the middle, creating a rotating whirlpool.

The most famous tidal whirlpool is the one that was once violent and dangerous in the Strait of Messina (between Sicily and the southern tip of the Italian mainland), which Homer depicted in his *Odyssey* as the second of two monsters, Scylla and Charybdis, faced by Ulysses. Another dangerous tidal whirlpool is the Malström in the narrow strait between two of the southern Lofoten Islands off Norway, which was written about by both Jules Verne and Edgar Allan Poe (Parker, 2012).

Nonlinear shallow-water effects on tidal currents

The shallow-water nonlinear processes that affect the tide (as discussed earlier) also affect the tidal currents, and in many situations those shallow-water effects are seen to be more dramatic in the tidal current than in the tide. The asymmetric tidal current can have (1) a shorter flood phase with higher speeds and a longer ebb phase with slower speeds (called flood dominance because of the higher flood current speeds), (2) a shorter ebb phase with higher speeds and a longer flood phase with slower speeds (ebb dominance), or (3) equal flood and ebb phases, but where one of the slacks can last for a couple of hours. Asymmetry in the tidal current is important in the transport of sediment and pollutants. The transport of coarse sediment depends on the maximum speeds achieved and so might be transported up an estuary with a flood-dominant situation. Fine sediment stays suspended except near slacks, so the case with longer slacks before ebb might lead to deposition at that time. (See Speer et al., 1991, for more discussion on flood and ebb dominance.)

The distortion in tidal currents can be greatly enhanced by a strictly linear superposition effect that, in fact, will not similarly enhance the distortion in the tide. An extreme example is in Ramshorn Creek, a small shallow channel connecting the shallow Cooper and New Rivers, both part of the Intracoastal Waterway in South Carolina and both connected to the Atlantic Ocean (Parker, 1991). Two tide waves, each distorted by shallow water, enter Ramshorn Creek at opposite ends and cross, superimposing their effects. The M_2 flood currents for the two waves are in opposite directions, and so the superposition of the two waves leads to a reduction in the M_2 tidal current. Each tide wave is distorted to a different degree, because the two waterways leading to Ramshorn Creek do not have exactly the same width or depth. Thus, in each waterway there is a different phase relationship between M_2 and M_4, so that when the M_4 tidal constituent waves are superimposed the two waves add together (rather than cancel each other out like the two M_2 constituent waves), thus increasing the M_4 tidal current. The result is the very unusual situation of a dominant quarter-diurnal tidal current, that is, the tidal current actually changes directions eight times a day instead of four (namely, there are four tidal current cycles per day instead of two, each cycle being 6.21 h long). There are other waterways in the salt marsh estuaries of South Carolina and Georgia that also exhibit the effects of two tide waves entering from opposite ends and crossing each other. Most are not as extreme as in Ramshorn Creek, but the result is still a very distorted tidal current curve. (This superposition effect does not lead to a dominant M_4 in the tide because the two M_2 tides being scalars add positively at the crossover point, not negatively as with the M_2 vector tidal currents.)

Summary

Although it is astronomical forcing (the gravitational effects of the moon and sun on the oceans) that makes the tides so predictable, it is the hydrodynamics (i.e., physics of the water movement) that determines the size and timing of the tides. When we speak of the tidal hydrodynamics of estuaries, we are speaking of shallow-water tides, which have much larger tidal ranges and faster tidal currents than open-ocean tides. In shallow water, the hydrodynamics also transfers tidal energy, through various nonlinear processes, to new frequencies. These so-called shallow-water tidal constituents include higher harmonics of particular astronomical constituents (called overtides), which distort the simple sinusoidal shape of the tide and tidal current curves that are found in the open ocean, often causing asymmetry in the tidal cycle in an estuary. These same nonlinear processes also lead to interactions between the tide and nontidal phenomena such as river discharge and wind-induced changes (e.g., storm surges). In this entry, we have explained how each of these mechanisms work and the importance of tidal hydrodynamics in moving the water and affecting the processes in estuaries.

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Cross-references

Mean Sea Level
Tidal and Nontidal Oscillations
Tidal Asymmetry
Tidal Datum
Tidal Hydrodynamics
Tidal Ranges
Tides

TIDAL AND NONTIDAL OSCILLATIONS

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Definition

Tidal oscillation is the oscillation in the vertical and horizontal movement of water (and the properties of water) caused by the tidal forces of the moon and sun.

Nontidal oscillation is the oscillation in the vertical and horizontal movement of water (and the properties of water) caused by nontidal phenomena that affect the estuary and ocean, such as wind, atmospheric pressure, air temperature, river runoff, and earthquakes.

Introduction

The movement of water into, out of, and inside an estuary is critically important for understanding all processes that take place in that estuary, whether they be physical, geological, chemical, or biological. There are numerous examples of water movement effects on such processes, only a few of which include transporting nutrients, flushing pollutants, moving sediments, moving floating ecosystems, changing the characteristics of water in which an ecosystem occurs, eroding shorelines, mixing water masses, and affecting stratification.

Oscillations

Many of the most important water motions involve some type of oscillation, where there is movement around some mean value (an equilibrium), that movement fluctuating between positive and then negative values around that mean. This can take many forms, such as the oscillation of the water surface up and down (around a mean sea level), water repeatedly flowing up and then down an estuary, water temperature increasing and then decreasing, salinity increasing and decreasing, as well as fluctuations in numerous chemical, biological, and geological characteristics affected by these physical oscillations. Properties in an estuary undergo oscillations caused by a number of the physical phenomena mentioned below.

Many of the most important of these oscillations are the result of some type of wave motion. A wave is essentially an oscillation that travels through space (in this case through water or along the water surface), transferring energy from one location to another and often transporting mass (there are mass oscillations, but with the nonlinear systems inherent in shallow estuaries, there is usually overall transport of mass and other properties after each complete oscillation). These waves can be of different frequencies (periods) and wavelengths, with different driving/restoring forces. Some of these waves (e.g., the long tide wave generated by the moon and sun in the open ocean) are *forced* waves, where their generation is continuous. Most other waves are *free* waves, where an initial generating force sets them in motion, to either travel as far as they can or cause oscillations in an estuary as long as they can, using that initial input of energy.

Long waves

The most powerful of these waves tend to be *long* waves, that is, waves whose wavelength (the distance from one peak to the next peak) is much greater than the water depth of the estuary (or for some long waves, even much greater than the depth of the ocean to which the estuary is connected). Most important are the tides, storm surges, nearshore wind waves, and tsunamis (although the wavelengths of nearshore wind waves are not nearly as long as for the other three types). Each of these long waves has a different generating force.

In their largest form, each of these long waves can be the cause of deadly *natural disasters* along the coast and inside estuaries. Although the easy and accurate prediction of tides makes tidal extremes less dangerous today, in the past large tides, tidal bores, and tidal whirlpools killed many people. Storm surges have killed many millions over the centuries, because they were not predictable and people living in low-lying coastal areas drown in the floods caused by the storm surges. That did not change until late in the twentieth century when satellites and hydrodynamic computer models allowed us to accurately predict where and when a storm surge would strike and then provide warnings to those in the danger area. Tsunamis are still not predictable (because the submarine earthquakes that cause them are not predictable), which is why we lost 300,000 people to a tsunami as recently as 2004 in countries around the Indian Ocean or 25,000 people even in a tsunami-knowledgeable country like Japan in 2011. Because of much better weather prediction, we can now predict when large wind waves are likely to strike the coast. For a history of marine disasters and a history of how scientists slowly developed a marine prediction capability, see Parker (2012).

In their less extreme form, tides, storm surges, and nearshore wind waves have very significant day-to-day effects on an estuary. They account for most of the water movement in the estuary (and its effect on biological and geological systems) along with river discharge (which could be thought of as having an annual asymmetric oscillation). Many of these long wave phenomena can happen at the same time, combining their effects. They also interact with each other because the shallow water causes nonlinearity in the motion. They do not just add to each other, they change each other, by transferring energy from one to another through various nonlinear mechanisms. (see *Tidal Hydrodynamics*, and also Parker, 2007, for explanations of mechanisms that cause the tide to interact with storm surges, wind waves, and river discharge).

Types of oscillations

In most estuaries, the *tides* provide the dominant forces that cause the water to move. The gravitational effects of the moon and sun on the oceans cause the water to move vertically (the *tide*) and horizontally (the *tidal current*), the entire phenomenon usually referred to as the tides. Tidal motion is actually the motion of extremely long waves generated in the open ocean, which propagate over the continental shelf and up into estuaries where their size is usually amplified. Only the oceans are large enough for the tide-generating forces to directly produce a tide of significant size. The tides in an estuary are forced at the estuary entrance by the tide wave from the ocean (having been modified to some degree by propagating over the shallower continental shelf). In estuaries we are essentially speaking of *shallow-water tides*, which are significantly different than open-ocean tides, having not only much larger tidal ranges and faster tidal currents but also distorted and asymmetric oscillations caused by the nonlinear effects of the shallow water, which can have significant effects on transport within the estuary (see *Tidal Hydrodynamics* and also Parker, 2007).

Storm surges are very long waves, generated by the fast winds of hurricanes and gales, which can push huge quantities of water onshore causing extensive flooding. Such wind-generated changes in water level along the shore also occur under more modest wind conditions (under conditions not considered as "storm" conditions), but the term storm surge is often used to define in general any change in water level caused by wind. Atmospheric pressure also plays a small role in storm-induced changes in water level. The crest of a storm surge does the most damage when arriving at the coast at high tide. Because of the shallow water, storm surges can modify the tide, sometimes decreasing the tide range when the storm surge has made the water depth greater (Parker, 2007).

Nearshore wind waves are also generated by wind, sometimes local winds, and sometimes by distant storms whose waves travel across the ocean, reaching distant shores as *swell*, which are deep-water waves until they reach the shallow waters near the coast and inside the estuary. At this point the bathymetry can focus wave energy via wave refraction, increasing their size at shallow points along the shore, creating longshore currents and rip currents, eroding shorelines, and mixing the water column. During storms, they do the most damage when arriving at the shore at high tide. Wind waves can also affect the tide by modifying the bottom friction that affects the propagation of the tide wave.

Tsunamis are very long waves generated most frequently by submarine earthquakes (ones that have significant vertical movement of the sea bottom) but also occasionally by volcanic eruptions or submarine landslides. Their wavelengths are very long in the deep ocean (hundreds of kilometers), where they also travel at great speeds (e.g., 600 km/h). When tsunamis reach the shallower continental shelf, their wavelength shortens, their speed decreases, their height increases, and they send large waves against the shore with periods usually on the order of 20–40 min. They do the most damage when arriving at the shore at high tide (Parker, 2012).

Beyond these four primary long-wave phenomena, there are other phenomena that can cause oscillations in an estuary. Examples are seiches, land breeze-sea breeze, and river flow.

Seiches are oscillations in harbors or small bays caused by wind waves at the harbor entrance. These oscillations are largest if the period of the waves entering the harbor matches the natural period of the harbor basin (as determined by its depth and width).

Land breeze-sea breeze oscillations in the wind are found in tropical regions and in temperate regions during the summer. During the day, the land heats up more than the sea and the air rises above the land to be replaced by cooler air blowing toward the land from the sea. The opposite breeze occurs at night when the sea is warmer than the land. A land breeze-sea breeze, which has roughly a 24-h period, can produce an oscillating wind current in the upper waters of the estuary or along the coast or even an oscillation in the water level (although these oscillations can be hidden by diurnal tidal oscillations). There are also changes in wind speed and direction as weather systems move over an estuary that can be quasiperiodic and can cause oscillations in the estuary.

The daily change in air temperature also produces a 24-h oscillation in the heating of the water surface, which affects the estuary as does the 24-h cycle in the effect of light on photosynthesis in phytoplankton. There is, of course, an annual (seasonal) oscillation in temperature (and other meteorological effects).

River flow can be thought of as having a very asymmetric oscillation, with most high flow values occurring after high spring runoff from melting snow or from rainfall produced by large storms or hurricanes, with the remaining time being characterized by much less flow. River currents augment the tidal current, making the ebb flow duration much longer and the flood duration much shorter. The river flow can also interact with the tide nonlinearly, reducing the tide range and distorting the shape of the tide curve (see *Tidal Hydrodynamics* and Parker, 2012).

Summary

Tidal oscillations are oscillations in the vertical and horizontal movement of water (and the properties of water) caused by the tidal forces of the moon and sun. Nontidal oscillations are horizontal and vertical oscillations in water and its properties caused by nontidal phenomena that affect the estuary and ocean, such as wind, atmospheric pressure, air temperature, river runoff, and earthquakes. This entry summarizes the effects of tides, storm surge, long wind waves, and tsunamis (all of which have long wavelengths), as well as other phenomena that can cause oscillations in an estuary, including seiches, land breeze-sea breeze, daily and annual changes in air temperature and light, and river discharge. Various aspects of the biological, chemical, geological, and physical environments of an estuary have equilibria around which there are oscillations caused by the above phenomena. Examples of these oscillations can be found elsewhere in this encyclopedia.

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Cross-references

Mean Sea Level
Seiche
Storm Surges
Tidal Hydrodynamics
Tides

TIDAL RANGES

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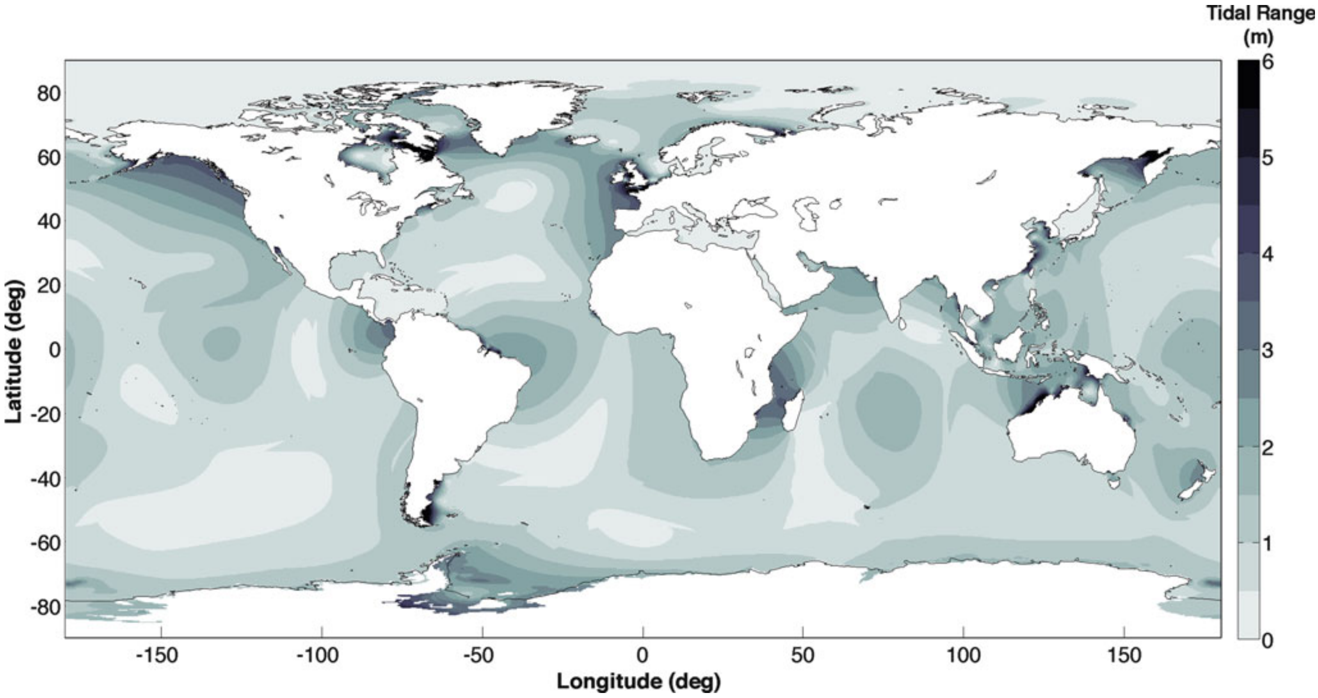
Definition

Tidal range is the vertical difference in height between consecutive high and low waters over a tidal cycle. The range of the *tide* varies between locations and also varies over a range of time scales (Stembridge, 1982). Differences in tidal range are important, as they are often related to variations in coastal processes and morphology (Davies, 1980).

Description

Temporally, tidal range varies due to changes in the position and alignment of the moon and sun relative to the earth. Every fortnight, the largest tidal ranges in semidiurnal regions occur during spring tides, when the moon and sun are in phase around times of new or full moon; while the smallest tidal ranges occur during neap tides, when the moon and sun are out of phase. In diurnal regions, the largest tidal ranges occur every fortnight during equatorial tides, when the moon is over the tropics; while the smallest tidal ranges occur during tropic tides, when the moon is over the equator. Over a month, tidal range changes as the moon moves from its closest (perigee) approach to the earth to its furthest approach (apogee) and back. Over annual time scales, changes in tidal range occur as the sun's position varies north or south of the equator and as it moves from its closest (perihelion) to furthest approach (aphelion) to the earth and back. The largest semidiurnal tidal range occurs in March and September during the equinoxes, while the largest diurnal tidal range occurs in June and December during the solstices. Over longer time scales, variations in tidal range arise as a result of the 8.85-year cycle of lunar perigee and the 18.61-year lunar nodal cycle (Haigh et al., 2011).



Tidal Ranges, Figure 1 Global distribution of mean tidal range (Estimated using tidal constituents derived from TPXO7.2 global ocean model (Egbert and Erofeeva, 2002) and downloaded from http://volkov.oce.orst.edu/tides/).

Spatially, tidal range varies according to the hydrodynamic response of a particular ocean basin, shelf sea, bay, or estuary to astronomical tidal forcing (Figure 1). Tidal ranges are typically smallest in the open ocean, along open ocean coastlines, and in almost fully enclosed seas, such as the Mediterranean. Conversely, tidal ranges are usually largest in semi-enclosed seas and funnel-shaped entrances of bays and estuaries, such as the Bay of Fundy in Canada or Bristol Channel in the UK, or regions where a continental shelf has the right combination of depth and width for tidal resonance to occur, such as on the northwest Australian shelf. Tides are often crudely classified by their mean range into macrotidal (>4 m), mesotidal (2–4 m), and microtidal (<2 m).

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Cross-references

Tidal Datum
Tides

TIDAL REFLECTION

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Synonyms

Tidal wave reflection

Definition

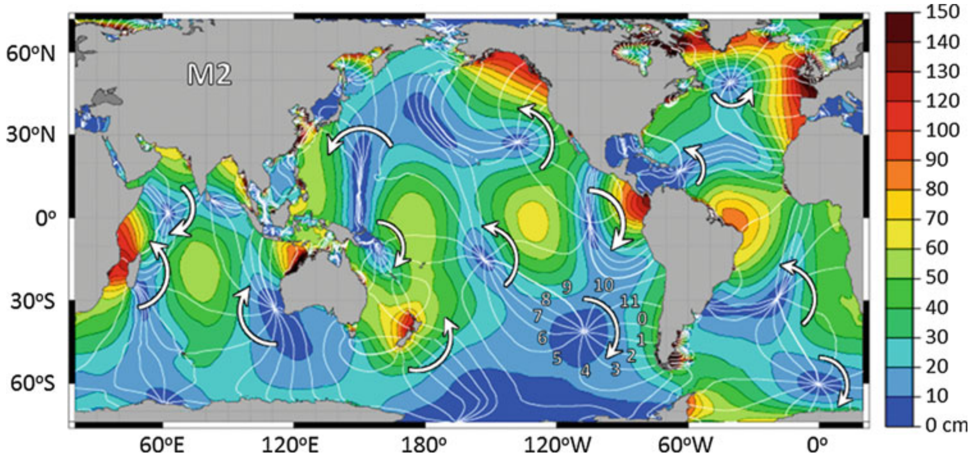
Tidal reflection is a transformation process experienced by a tidal wave in which all or part of its energy bounces back due to geometric variations in the boundaries.

Description

Reflection is a common phenomenon experienced by waves, and water waves are not an exception. Tidal wave reflection occurs in a wide range of spatial scales, ranging from tens of meters to thousands of kilometers. Semi-enclosed bodies of water, such as estuaries, allow (co-oscillating) tides to enter through their open boundaries. As the tidal wave propagates upstream, bathymetric and topographic changes and the presence of obstacles bounce back all or part of the incoming tidal energy. Prominent tidal reflections typically occur at abrupt channel constrictions and at upstream dams and weirs (e.g., in the Guadalquivir estuary in Spain and in the Ems in the Netherlands). Tidal reflection changes the structure of tidal currents and ranges, which in turn affects the dynamics of sediments and other solutes, and may also generate residual movements with consequences in the long-term morphology, water quality, and biota.

The prototypical case of tidal reflection is that of a monochromatic tidal wave propagating into a straight, frictionless channel of constant depth and width that is closed at its head with an impermeable barrier. When the traveling tidal wave encounters the head, all its energy is reflected back as a sinusoidal wave of the same amplitude and frequency that propagates in the opposite direction. Physically, the oscillation resulting from their superposition is a standing wave with an antinode at the head, i.e., a position where the current vanishes and the amplitude is twice the amplitude of the incoming wave. If the semi-enclosed body is long enough in comparison to tidal wavelength, at even/odd multiples of a quarter of the wavelength off the head, antinodes/nodes are observed. A tidal node is a location where the free surface does not vary (null tidal elevation), and maximum along-channel current amplitude occurs. Because the wave length depends on the tidal period, different constituents will exhibit different nodal point locations. For standing waves, the tidal elevation and the current velocity are $\pi/2$ out of phase. In fact, the character of the tidal wave is usually analyzed (although not uniquely characterized) by comparing the phase lag between the high water and the high water slack with the values that would be obtained under ideal conditions for standing waves (0) and progressive waves (T/4), where T is the tidal period. Resonance phenomena are also possible and not infrequent in estuaries. For a given constituent, resonance may occur if the estuary length L is about a quarter wavelength of the forcing tide or odd multiples of thereof. This means that the ratio of the maximum tide at the closed end to the tide at the entrance becomes large. Perhaps the most dramatic example of quarter-wave resonance is that observed for the M2 semidiurnal tidal constituent in the Bay of Fundy (Canada).

Friction dampens both the incident and reflected wave; modifies their wave numbers, which depend on the along-channel position; and also moves the position of nodes and antinodes. If dissipation is significant and the estuary is long enough, tidal energy may not reach the closed end and no reflected wave is generated there. Even though the incoming wave arrives with sufficient energy to the closed end, the effective impact of tidal reflection may be limited to a fraction of the estuary. The region of influence of tidal reflection can be delimited by the position at which the ratio between the amplitudes of the reflected and the incident waves is small. This practical approach requires separating the incident and the reflected wave by means of



Tidal Reflection, Figure 1 Worldwide amphidromic systems for the semidiurnal M2 tide (12.42 h) obtained from Topex/Poseidon satellite observations. *Color contours* indicate co-range areas in cm, and *white lines* are co-tidal lines. The amphidromic point east of Chile coast shows the time of high water at each hour in the tidal cycle. *Arrows* show the clockwise or anticlockwise character of several amphidromic points (Image adapted from http://svs.gsfc.nasa.gov/stories/topex/index.html by Richard Ray, Space Geodesy branch, NASA/Goddard Space Flight Center).

analytical and observational methods. Tidal phases easily obtained from harmonic analysis also provide useful information because they tend to flatten out near the reflector (different positions reach high water or low water at the same time) when reflection dominates over friction. The wave number obtained directly from the observed tidal phases is thus typically much smaller than the wave numbers of the incident and reflected waves. Frictional effects can significantly change the resonance condition and the maximum amplification. Nevertheless, due to complex bathymetry, and the variety of forcings involved, a precise estimate often requires computational techniques.

In many natural situations, reflection may also introduce a phase shift between the incoming and reflected waves that affects the position of the antinodes and nodes. Moreover, only a fraction of the incoming wave is reflected. Energy reflection is characterized by a (complex) reflection coefficient R, which is simply the ratio between reflected and incident wave amplitudes. Due to the imperfect reflection ($R < 1$), there are no true nodes or antinodes in the wave profile, but quasi-antinodes (maxima of the tidal elevation) and quasi-nodes (minima of the tidal elevation). In simple terms, the resulting wave is a partially (instead of fully) standing wave, made up of a combination of a standing wave plus an incoming progressive wave. In general, the reflection coefficient is frequency dependent and varies spatially with the absolute distance to the head due to the increase in the dissipation and inverse shoaling of the reflected wave. Nonlinear interactions between the two waves and with other constituents may also be significant.

Tidal reflection also occurs due to topographic changes, gradually as the wave propagates (e.g., variations in depth, channel width, presence of meanders with small radius of curvature, etc.). If the length scale of the spatial changes in section is much larger than the tidal wavelength, energy losses due to reflection are negligible and Green's law, which simply relates changes in channel width and depth to changes in tidal range, is applicable. However, this occurs rarely. In most cases tidal reflection is important (e.g., at the sill of a fjord), and Green's law is for practical purposes inapplicable. Wave reflection from the margins is also expected due to the geometry variations. For instance, channel convergence causes partial reflection of the incoming wave, thereby losing energy (apart from bottom friction) as it propagates upstream. The seaward radiation of (reflected) tidal energy due to channel convergence and, if any, reflection at the upper limit may modify the ocean tide near the basin entrance, especially when the basin is close to resonance.

The basic tidal movement in the open ocean is also a system of standing wave oscillations. Oceanic boundaries (mainly continents) reflect part of the tidal wave energy, setting up nodal points, and also resonant motions, if conditions exist for them. This reflection is responsible for the differences of timing and tidal range along the coasts, which in turn influence to a greater extent the estuary mouth morphology (e.g., wave- or river-dominated embayments occur where tidal action is weak). The constraining effects of the emerged lands and the influence of the Coriolis force result in the development of amphidromic systems (Figure 1), in each of which the tidal wave propagates around each tidal cycle. The range of the rotating tide depends on the distance from the amphidromic point: the larger the distance, the higher the range. Co-range lines join places with the same tidal range. These lines form more or less concentric circles around an amphidromic point. Co-tidal lines, which link all the points where the tide is at the same tidal phase, radiate from the amphidromic point. Tidal waves in amphidromic systems are typically Kelvin waves, namely, gravity waves influenced by earth rotation. They propagate along the coast, leaving it to the right (left) in the northern (southern) hemisphere when facing in the direction of wave propagation. Their tidal ranges decrease exponentially with increasing distance to the coast, and currents are in geostrophic balance in the direction perpendicular to the propagation. With a few exceptions, the tidal waves of amphidromic systems tend to rotate anticlockwise in the northern hemisphere and clockwise in the southern hemisphere. Again, local resonance is possible if the dimensions of the basin match the tidal wavelength. For instance, the North Sea provides a good example of a resonant amphidromic system.

Summary

Tidal wave reflection is a quite common transformation process experienced by a tidal wave and occurs simultaneously in a wide range of spatial scales. Sometimes the reflection is prominent (in terms of fraction of reflected energy), as for instance at a head weir, but sometimes is subtle, as occurs when tides propagate into a long, weakly convergent estuary. This phenomenon can even change the wave nature. Tidal reflection may generate residual movements which in turn influence to a greater extent long-term estuarine morphology and water quality.

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TIDES

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Definition

Tides are the regular and predictable rise and fall of the sea caused by the gravitational attraction and rotation of the earth, moon, and sun system.

Introduction

Tides are the regular and predictable rise and fall of the sea caused by astronomical forcing. The study of tides has a long history and is perhaps the oldest branch of physical oceanography (Cartwright, 1999). One of the fascinations of tides is the number and diversity of the practical and scientific applications (Pugh, 1987). These include concern with problems of navigation, coastal flooding and erosion, transport of pollutants, and the extraction of tidal power. Tides also have a controlling influence on geological, sediment transport, water quality, and marine biological processes.

Tides are normally used to refer to the vertical change in sea level, whereas the term *tidal currents* is used for the horizontal movement of water (Parker, 2005). Sea level relates to the vertical change in the height of the sea surface which occurs over all time and space scales from many different mechanisms (including *waves, seiches, storm surges, tsunamis*), with tides being the most predictable and the dominant component of sea-level variability in many parts of the world's oceans and coasts. Water level is the height of the sea surface above some reference level or benchmark, often called a *tidal datum. Mean sea level* is the average height of the sea over longer periods of time (usually a month or year), and hence the shorter-term variations of the tide are filtered out.

Tide-generating force

Newton's laws of gravitation attraction and motion form the basis for physical understanding of how tides are generated. The universal law of gravitational attraction states that the force (F) of attraction of two particles of masses M_1 and M_2 separated by distance R is

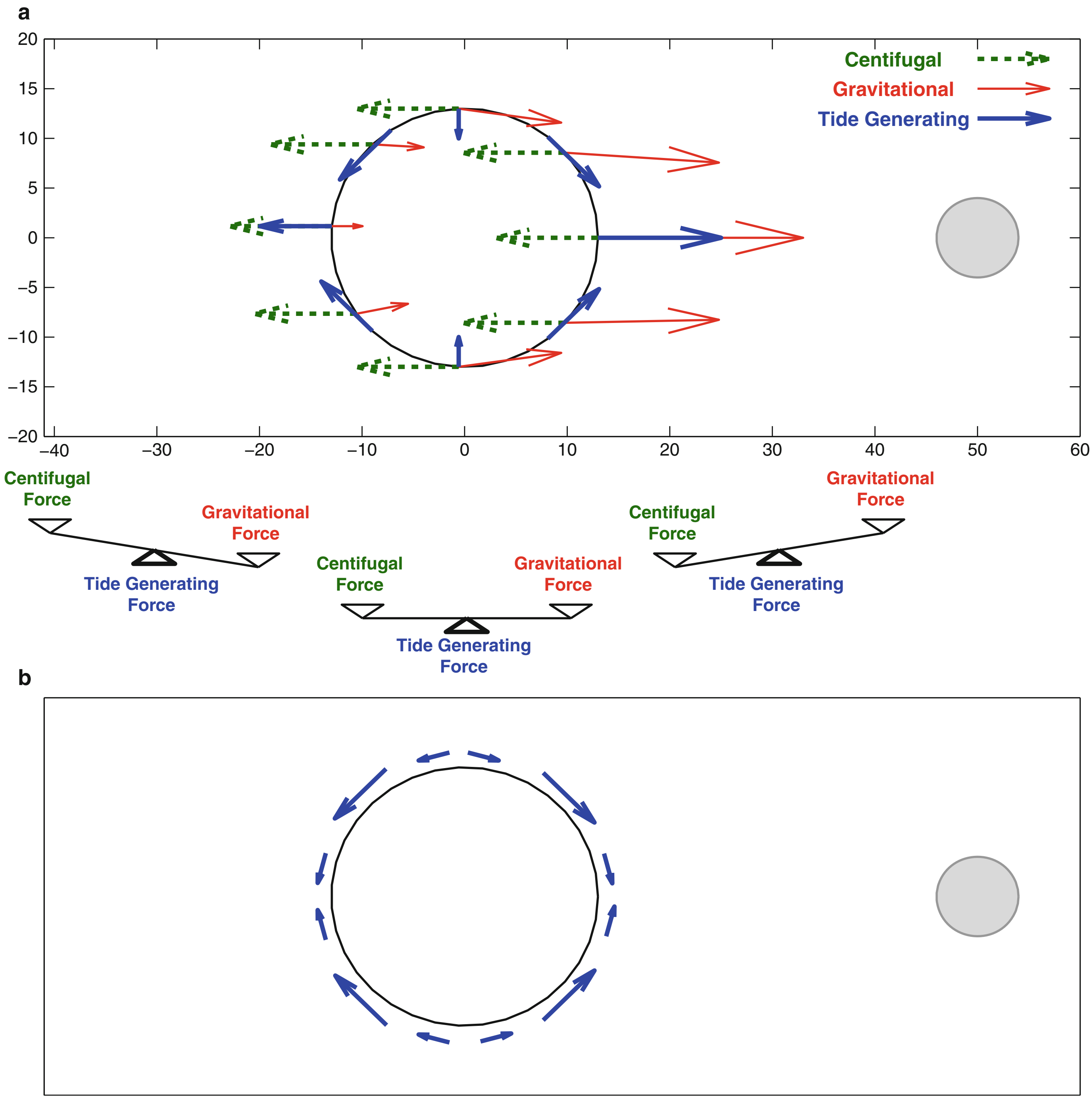
$$F = G\frac{M_1 M_2}{R^2} \tag{1}$$

where G is the universal gravitational constant. Newton's laws of motion assert that a body (e.g., an element of seawater) remains at rest or constant velocity, unless acted upon by a force, and that force is the product of a body's mass and acceleration.

To begin, consideration is given to the moon's role in generating tides. The earth and moon form a single system, mutually revolving around a common center of mass (known as the barycenter), with a period of 27.32 days. The earth is 81 times larger than the moon, and as a result the location of the barycenter lies within the earth. The earth and moon circle about the barycenter, without any rotation in absolute space. Therefore, each point on the earth travels in circles which have the same radius. Hence, they experience an equal centrifugal force (F_c), directed parallel to a line joining the centers of the earth and moon. While the centrifugal force is the same everywhere on the earth, the gravitational force varies with location and is directed toward the center of the moon. The local gravitation force (F_g) is given by

$$F_g = G\frac{M_e M_m}{(R-r)^2} \tag{2}$$

where M_e and M_m are the mass of the earth and moon, respectively, and r is the distance between the earth's



Tides, Figure 1 (**a**) The magnitude and direction of the tide-generating force on the earth (looking side on to the earth), created by the balance between the centrifugal force and the gravitational force. (**b**) The horizontal component of the tide-generating force which causes water movement.

center and the point of interest. Locations nearer the moon experience a local gravitational force that is larger than that experienced at locations farther away.

For the earth-moon system to remain in equilibrium, the total centrifugal force must exactly balance the force of gravitation attraction, or else the earth and moon would accelerate away or toward each other. At the center of the earth the two forces balance, but at locations nearer to the moon the gravitation force is larger than the centrifugal force, and for locations farther away, the centrifugal force is larger than the gravitational force (Figure 1a). This difference is the tide-generating force (F_t), which at a point on the earth is the difference between (1) and (2), which simplifies to

$$F_t = Ga\frac{M_e M_m}{R^3} \tag{3}$$

where a is the radius of the earth.

The local variation in the tide-generating force does not cause the water on the earth's surface to be drawn into two bulges on opposite sides of the earth. Instead it is the small horizontal component of the force which causes water movement (Figure 1b). This force, although small, has nothing to oppose it; whereas the vertical component of the tide-generating force acts against the much larger gravitational attraction acting upon the ocean.

The solar tidal force can be thought of in a similar manner. The mass of the sun is much greater than the mass of the moon, but this is offset by the greater distance from the earth. Therefore, the solar tidal forces are a factor of 0.46 weaker than the lunar forces.

Equilibrium tidal theory

To improve understanding of tides, Newton developed the equilibrium tidal theory, in which three main assumptions are made: (1) the earth is covered by an ocean of uniform depth with no land masses; (2) the ocean responds immediately to the tide-generating force; and (3) the effects of rotation and friction can be ignored (Masselink et al., 2003). Let us assume a stationary moon, aligned with the earth's equator. The lunar tide-generating force would cause bulges on opposite sides of the earth in our uniformly deep ocean (Figure 2a). The earth rotates in an anticlockwise direction on its polar axis, beneath these bulges, taking 24 h (a solar day) to complete one rotation. Hence, anywhere on the earth's surface you would experience two high tides each day, but with decreased height as you moved north or south of the equator (Figure 2b). In reality by the time the earth has completed one full rotation, the moon has moved on in its rotation around the barycenter. It takes about 50 min longer than a solar day for the point on the equator to reach the maximum part of the first bulge again. This is why high semidiurnal tides occur every 12 h and 25 min, not every 12 h.

At any point on the earth, the change in the height of the tide can be described by a simple harmonic curve of the form (Figure 2b, d)

$$H\cos(\omega t - g) \tag{4}$$

where H is the amplitude, ω is the frequency, g is the phase lag relative to a defined time zero, and t is time. Assuming that the moon rotates around the earth in a purely circular path aligned with the equator, we term this harmonic M_2. If we were to ignore the moon and just consider the sun, then the same features would be observed, except the time between high waters would be exactly 12 h and the height of high water would be just less than half of that observed for the moon. This harmonic is termed S_2.

Now consider the combined effect of the moon and sun. When the earth, moon, and sun are aligned during full or new moon, the equilibrium tidal bulges associated with the moon and sun are in phase and combine to create a spring tide with a larger than average tidal range (Figure 3). This occurs about every 15 days. When the moon is positioned half way between the new and full phases, the two tidal bulges are out of phase and combine to produce a neap tide with smaller than average tidal range.

Tidal patterns are complicated by the fact that for much of the year the moon and sun are not aligned with the equator. The earth rotates around the sun on a plane called the ecliptic, which is inclined at 23°27′ to the equator. The moon orbits the earth on a plane that is included by 5°9′ to the ecliptic. As the moon revolves, its position above the earth varies between latitudes 28°36′ (23° 27′ + 5°9′) north and south of the equator, and the tidal bulges are titled relative to this angle (Figure 2c). An observer on the earth's surface would experience two high tides a day, but the relative heights of each would be different, with the relative differences increasing moving farther north or south (Figure 2d). This so-called diurnal inequality reaches a maximum when the moon is at its maximum northern or southern declination about every 2 weeks.

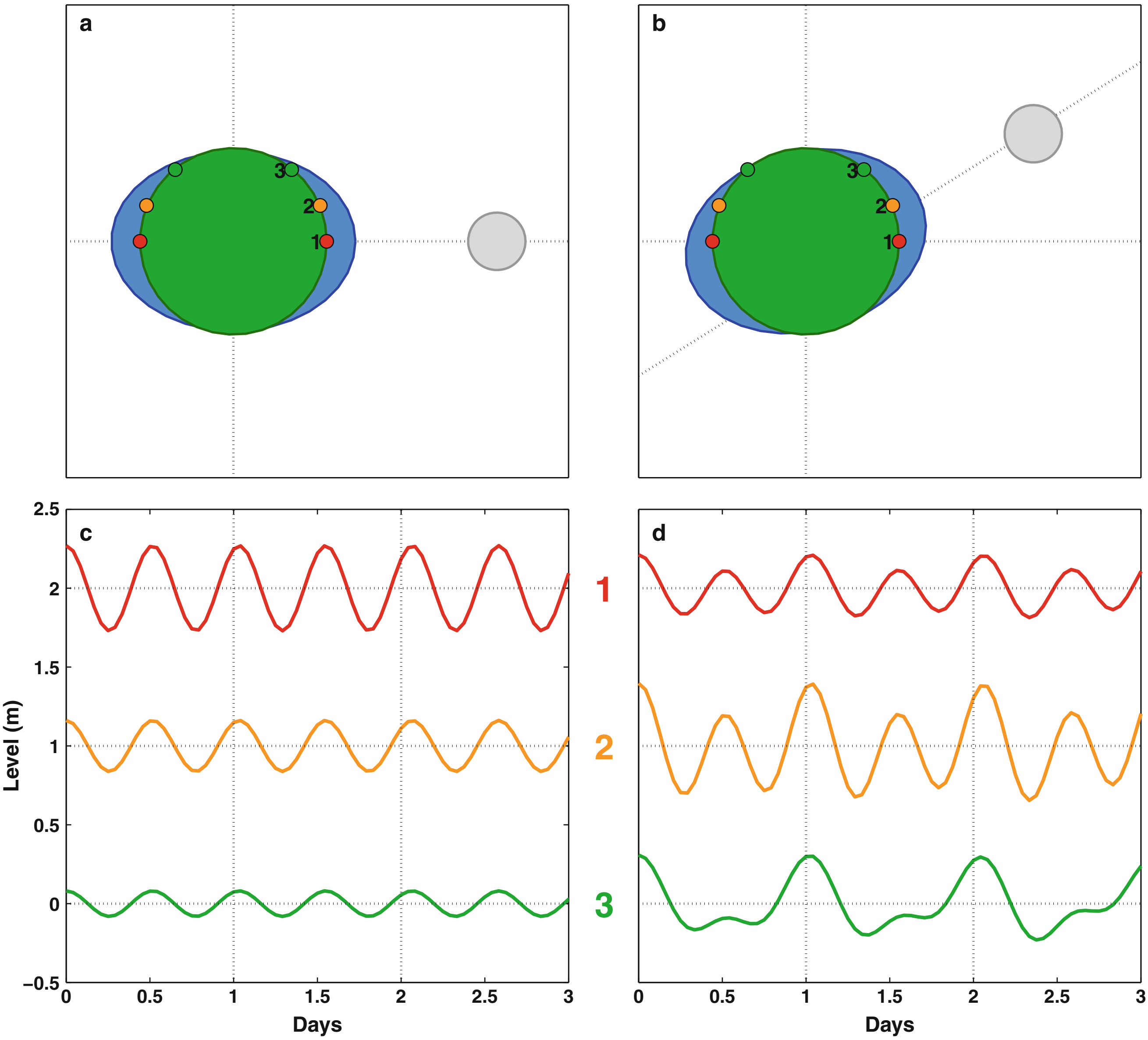
The sun's declination varies over a 365.25-day cycle, by 23°27′ north or south of the equator, causing the seasons. The largest diurnal tides occur in June and December during the solstices and the smallest diurnal tides in March and September during the equinoxes. As the strength of the diurnal force increases, the semidiurnal force reduces, and vice versa (Pugh, 2004). Hence, the semidiurnal tidal forces reach their maximum at the equinoxes.

Tidal patterns are further complicated because the orbits of the earth and moon are elliptical, not circular. As a result the distance between the moon and sun with the earth varies, altering the strength of the tide-generating force. Over a period of 27.6 days, the moon moves from perigee (its closest approach to the earth) to apogee (its farthest approach to the earth) and back. At perigee and apogee the lunar tidal forces are 15 % greater or less than average, respectively. The orbit of the earth around the sun is also elliptical, moving from perihelion (closest to the sun) to aphelion (farthest from the sun) over a period of 365.25 days. The difference in distance between these two positions is only about 4 % so tides are only marginally larger at perihelion compared to aphelion.

Over longer time scales, variations in tidal forces arise as a result of the 8.85-year cycle of lunar perigee (which influences tides as a quasi 4.4-year cycle) and the 18.61-year lunar nodal cycle (Haigh et al., 2011).

Harmonic analysis and tidal prediction

Predicting times and heights of tides has many practical applications, and there has been a long history of development from simple techniques that related the time of high

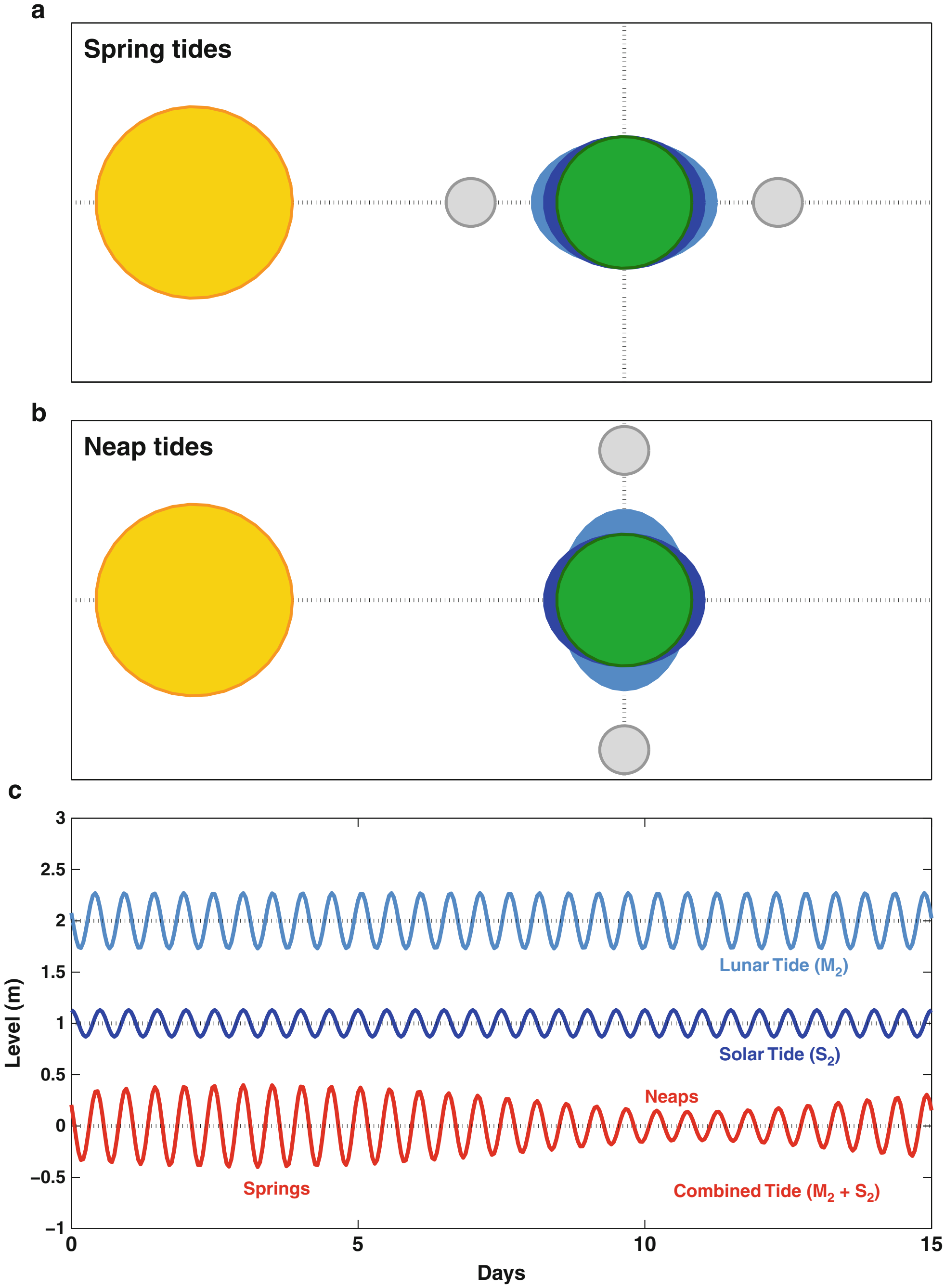


Tides, Figure 2 (**a**, **b**) The earth's (looking side on to the earth) double tidal bulge aligned with the moon which is located over the equator in (**a**) and over the Tropic of Cancer in (**b**). (**c**, **d**) Tidal curves (offset for presentation purposes) over a three-day period at sites 1, 2, and 3, the locations of which are marked on plots (**a**) and (**b**), respectively.

water to the phase of the moon (Pugh, 1987), through to the more sophisticated harmonic (Doodson, 1921; Foreman, 1977) and response analysis methods (Munk and Cartwright, 1966). Harmonic analysis is the most widely used tidal prediction method. The underlying principle of this method is the assumption that no matter how complex they appear, tidal variations at any location can be represented by the sum of a finite number of harmonic terms (tidal constituents), each expressed in the form of (4).

The frequency of each of the tidal constituents can be expressed as a linear superposition of six fundamental astronomical forcing harmonics, from which the positions of the sun or moon, and hence the tide-generating forces, can be calculated for any time. Therefore, it is possible to relate each individual tidal harmonic to real astronomical behavior (Pugh, 2004). More than 400 tidal constituents have been identified (Cartwright and Tayler, 1971), but the tide can be predicted to a useful level of accuracy using only a small subset of these.

In order to predict tides at a given location, we need to know the frequencies, amplitudes, and phases of the tidal constituents. The frequencies are fixed, but the amplitudes and phases of the different constituents vary with location and the aim of the harmonic analysis is to determine them.

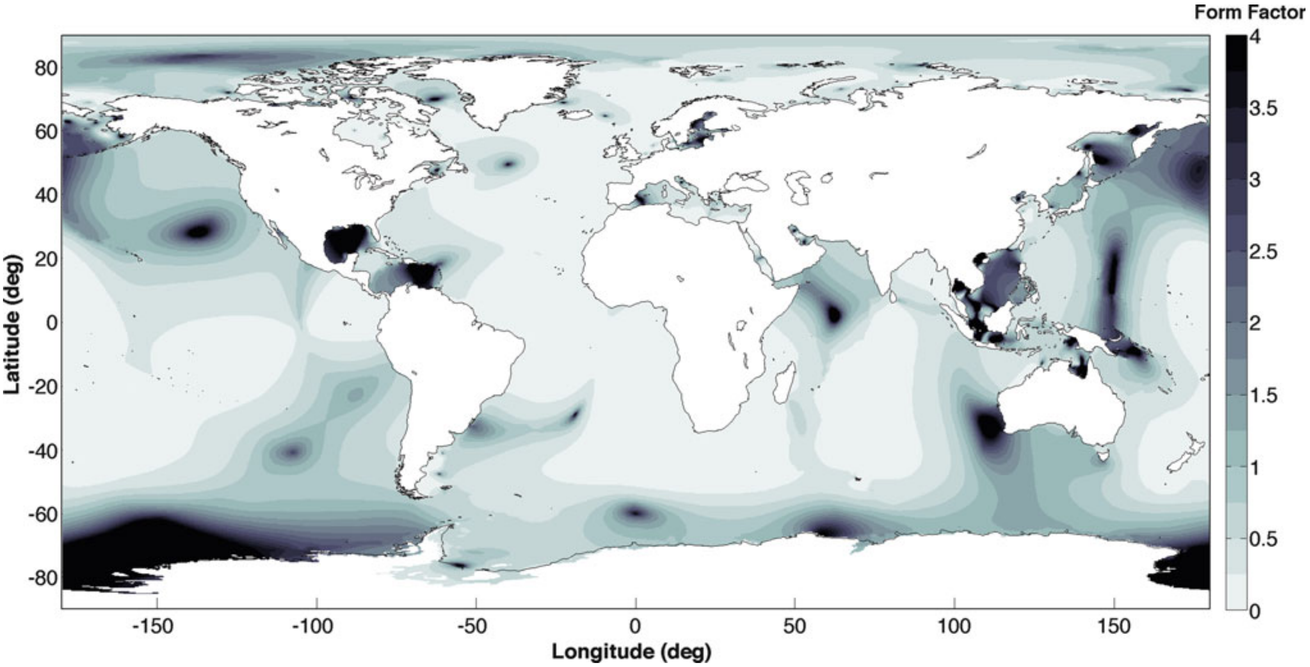


Tides, Figure 3 View looking down upon the earth from the north pole when (**a**) the earth, moon, and sun are aligned (during full or new moon) – the equilibrium tidal bulges associated with the moon and sun are in phase and combine to create a spring tide; (**b**) the moon is positioned half way between the new and full phases – the two tidal bulges are out of phase and combine to produce a neap tide with smaller than average tidal range. (**c**) Lunar, solar, and combined tidal curves (offset for presentation purposes) over a 15-day period at a point on the equator.

This involves least-squares fitting of the tidal constituents to water level measurements at that site. To predict the tides into the future, the estimated amplitudes and phases of each constituent can be substituted into (4) and then the terms summed.

Dynamic tidal theory

The equilibrium theory helps to describe several features of tides and serves as a reference system for harmonic analysis. However, the tide observed in reality bears no resemblance to the equilibrium tide, because none of the



Tides, Figure 4 Global distribution of the tidal form factor (Estimated using tidal constituents derived from TPXO7.2 global ocean model (Egbert and Erofeeva, 2002) and downloaded from http://volkov.oce.orst.edu/tides/).

principle assumptions of the theory are valid (Brown et al., 1989), including the following: (1) the presence of land prevents tidal bulges from directly circumnavigating the globe; (2) the rotation of the earth on its polar axis is too rapid for the inertia of the water masses to overcome in sufficient time to establish an immediate equilibrium tide; and (3) water movements induced by tide-generating forces are subject to friction and the Coriolis force.

The combination of all of these factors results in the development of so-called amphidromic systems. The global ocean is essentially broken into separate basins separated by shallow continental shelves and landmasses (Masselink et al., 2003). In these enclosed basins, the tidal wave travels from east to west, elevating sea levels against the western margin of the basin. The resulting slope in the ocean surface produces a pressure gradient that causes water to flow eastward. In the Northern Hemisphere, the eastward flow is deflected to the right by Coriolis force, which causes elevated sea levels against the southern margin. The pressure gradient force subsequently drives the water northward which is then deflected to the west and so on. Thus, the wave crest rotates around the ocean basin in an anticlockwise direction, which is referred to as a Kelvin wave. Amphidromic systems are set up on continental shelves and large bays, but these systems are scaled down and are less symmetrical as the wave speeds decrease in shallow water where there are significant energy losses due to bottom friction (Pugh, 2004).

In the open ocean, tides are generated directly by the tide-generating forces. These waves then spread onto the surrounding shallow shelf seas where their characteristics are altered through standing wave generation and local resonances (Pugh, 2004). As the tidal wave progresses from the deep ocean onto the shallow continental shelf, the wave slows down. The wavelength is reduced and the amplitude increases because the wave energy becomes concentrated in a smaller area. As the wave approaches even shallower water near the coast, it encounters the irregular topography of the coastline and moves into estuaries and bays where it undergoes even larger distortions. As the tide travels into a narrowing bay or estuary, the tidal range increases again, because the same energy is being forced through a smaller opening (Parker, 2005). In extreme cases, this can result in the formation of a tidal bore. In very shallow water, the wave crest will travel faster than the trough, resulting in *tidal asymmetry*. Eventually, the tidal amplitude will be reduced by bottom friction as the wave progresses into very shallow water.

Tidal characteristics and types

At a particular location, tides are typically classified by their range (see *Tidal Ranges*) and form. The form of the tide can be estimated from the tidal form factor (F). This is derived from the main diurnal and semidiurnal tidal constituent amplitudes (H) using the following equation (Pugh, 2004):

$$F = \frac{(H_{K_1} + H_{O_2})}{H_{M_2} + H_{S_2}} \tag{5}$$

It can be used to quantify whether a site experiences (Figure 4) the following: two high and low tides each lunar day, semidiurnal tidal form ($F < 0.25$); a single high and low tide, diurnal tidal form ($F > 3$); or periods when both types occur, mixed tidal form (mixed, mainly semidiurnal, $F = 0.25$ to 1.50; mixed, mainly diurnal, $F = 1.50$ to 3.00).

Summary

To fully understand tides and predict them, one must understand both the astronomical forcing that gives rise to them and also the response of the oceans, bays, and estuaries to this forcing (Parker, 2005). Knowledge of the astronomical forcing helps to describe several features of tides and servers as a basis for the tide's predictability. However, it is the knowledge of the hydrodynamics of the tide in a particular ocean basin, shelf sea, or estuary which provides understanding of the timing of high and low water, the size of the tidal range, the form of the tide, and any complex shallow water distortions of the tidal curve.

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Cross-references

Seiche
Storm Surges
Tidal and Nontidal Oscillations
Tidal Asymmetry
Tidal Datum
Tidal Hydrodynamics
Tidal Ranges

TIMESCALE

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Definition

Time scale refers to "the time allowed for or taken by a process or sequence of events" (OUP, 2013).

Essential concepts, application and usage

The above definition provides a general description of the term "time scale." In science and engineering practice, a time scale is typically an *estimate* expressing a representative, overall magnitude as opposed to a precise value; as such, time scales (like length, velocity, and other commonly used scales) are frequently quantified in terms of orders of magnitude (Cushman-Roisin and Beckers, 2011). A time scale carries the dimension of time and may be expressed either in precise temporal units (e.g., seconds, hours, days, years, decades) or in approximate temporal units (as with "tidal," "seasonal," or "episodic" time scales).

In combination with other scales, time scales may be used to estimate the relative magnitude (and thus, importance) of individual terms in time-marching equations (Cushman-Roisin and Beckers, 2011). Time scales may be defined to describe the rates of physical, biological, and chemical processes, serving as a common currency with which to compare their relative speed, and thus importance (e.g., Koseff et al., 1993; Middleburg and Nieuwenhuize, 2000; Lucas et al., 2009; Dame, 2012). In such a context, a smaller time scale suggests a faster and more dominant process (Lucas, 2010). A time scale may be estimated as the reciprocal of a first order rate constant or frequency, or from a combination of length, velocity, diffusivity, or other scales such that the remaining dimension is *time* (e.g., Fischer et al., 1979; Koseff et al., 1993). A time scale may reflect the approximate time for completion of a process, such as for (1) diffusive mixing over an estuary's depth (Fischer et al., 1979), (2) advective transport over the estuary length (MacCready and Banas, 2011), (3) flushing of an estuary by river flow and/or tides (Sheldon and Alber, 2006), (4) settling of particles through a water column (de Brauwere and Deleersnijder, 2010), (5) biomass growth by a factor *e* (Lucas, 2010), or (6) filtration of a water column or estuary volume by benthic organisms (Buzzelli et al., 2013).

The term "time scale" may also be used to convey (1) a typical period of variability or fluctuation in estuarine forcing or ecosystem response (e.g., Litaker et al., 1993; Cloern, 1996; Jay et al., 2000), (2) a period of estuary adjustment to low-frequency forcing (e.g., Monismith et al., 2002; MacCready and Geyer, 2010), (3) the period(s) of variability captured by measurements or models (e.g., Blumberg et al., 1999; Wang et al., 1999), or (4) the temporal lens through which processes are examined (e.g., Stacey et al., 2001; Chapin III et al., 2006).

Summary

Described very generally, a "time scale" is the time taken by a process or sequence of events (OUP, 2013). However, there exist many shades of meaning and usage of this term, which is ubiquitous in engineering and increasingly so in environmental science. Typically, a "time scale" implies a time *estimate*, not a precise value. Comparison of time scales for two or more coincident processes (be they physical, biological, or chemical) provides a diagnostic tool for assessing the processes' relative importance and, potentially, for simplifying complex equations.

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Cross-references

Residence Time

TOMBOLO

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Synonyms

Ayr (Northern Europe); Tie bar

Definition

A tombolo is a sediment deposit at the coast formed by wave refraction and diffraction at the edges of an obstacle (natural or artificial) originally detached from the mainland.

Description

The formation of a tombolo is caused by the diffraction of the wave at the ends of an obstacle, creating a convergence of opposing flows of sediment transport. Depending on the input of sediments and sea-level trends, the accumulation of sediments (sands or gravels) leads to continuous progradation of the coastline towards the detached obstacle (usually an island). A smaller sediment protrusion can also be developed in the back side of the obstacle. The final phase of this process is the complete coalescence between these sediment features and the blockage of the water body located behind the obstacle, with the consequent formation of a connection between the obstacle and the mainland (i.e., the tombolo). Tombolos can be classified as simple, when the feature is formed by a single sediment body, or complex (doubles, triples, etc.), in which the sediment ridges can be formed in a different time (Johnson and Reed, 1910). In this last case, lagoons can be formed between ridges, marking intervals of interruption of the tombolo formation.

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Cross-references

Coastal Lagoons
Coastal Landforms
Shoreline

TOXIC BLOOMS

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Synonyms

Allelopathic algae; Harmful algal blooms (HABs); High-biomass blooms; "Red tides"

Definition

Toxic blooms are the accumulation of toxin-producing cells that lead to aquatic living resource mortalities or illness or deaths in avian or terrestrial mammals, including humans.

Mortalities from blooms can also occur through indirect effects of the accumulation of high biomass typical of blooms and the subsequent respiration or death and decomposition that consumes the available oxygen, leading to suffocation and dying and dead fauna.

Introduction

Toxin-producing taxa and syndromes

Toxic blooms are generally accumulations of phytoplankton (microscopic single-cell free-floating plants) and occasionally mixotrophic or heterotrophic dinoflagellates, benthic dinoflagellates, or macroalgae. Some of the cells accumulating produce complex intracellular compounds that are toxic to various freshwater and marine fauna as well as higher organisms that include birds and mammals. The suite of toxins produced (http://www.issha.org/Welcome-to-ISSHA/Harmful-Algae-Links) has a primary toxin and then many substituted congeners and include paralytic shellfish poisons (PSP), primarily from saxitoxin and gonyautoxins and their derivatives in marine dinoflagellates like *Alexandrium* species and others; neurotoxic shellfish poisons (NSP) such as brevetoxin produced by *Karenia brevis* in Florida red tides and anatoxin from some freshwater cyanobacteria; domoic acid for amnesic shellfish poisons (ASP) produced by some marine members of the diatom genus *Pseudo-nitzschia*; ciquatera fish poisons (CFP) from toxin precursors for ciguatoxins and maitotoxins produced by some benthic marine dinoflagellate species like *Gambierdiscus*; diarrhetic shellfish poisons (DSP) from okadaic acid produced by the marine dinoflagellate *Dinophysis* and ichthyotoxic karlotoxins from the brackish dinoflagellate *Karlodinium*, which kill many fish and dramatically reduce grazing pressures from small planktonic grazers; azaspiracids which induce vomiting and diarrhea, produced from the marine dinoflagellate *Azadinium spinosum*; and the hepatotoxic microcystins produced by *Microcystis* and other freshwater cyanobacteria. Other toxins (prymnesins, nodularin, yessotoxin, palytoxins, tetrodotoxin, spirolides) are also found less frequently throughout the world. Exposure for most birds and mammals is primarily through shellfish and other organisms that concentrate toxic cells and are subsequently ingested by the larger consumers, including humans; drinking water supplies may also be toxin sources for some of the freshwater toxins. Distributions of these toxins and bloom formers can be viewed at http://www.whoi.edu/redtide/regions/world-distribution.

High-biomass blooms

Toxin- and nontoxin-producing populations of the plankton and macroalgae noted above can accumulate to such high levels that the oxygen demand associated with the very abundant organisms can consume all available oxygen in surrounding waters, posing serious threats to aerobic populations of fish, shellfish, and many benthic taxa. These high-biomass blooms are increasingly common to estuaries, and shallow coastal lagoons receiving high nutrient loads (see Anderson et al., 2002; Heisler et al., 2008) and bloom formers include *Microcystis* and *Anabaena*, *Prorocentrum*, *Ceratium*, *Cochlodinium polykrikoides*, raphidophytes, pelagophytes (brown tides), and other HAB species. Dense beds or scoured macroalgae can also occur, leading to high respiratory demand during decomposition and oxygen-poor conditions that threaten coexisting fauna. Finally, spring diatom blooms can be very large in temperate estuaries where post-bloom deposition and decomposition lead to hypoxia or anoxia in bottom waters (e.g., Chesapeake Bay, Kemp and Boynthon, 1992); Asian countries typically include diatoms in lists of HABs from regional waters.

Toxicity from other compounds (allelopathy)

Many of the bloom-forming species produce other compounds that are toxic to other members of the plankton, whether other phytoplankton taxa that compete for nutrients and sunlight in active growth or potential grazers that could limit accumulation. These compounds are "toxic" in that they provide the bloom former greater opportunity for maintaining growth, while their competitors and predators are limited in their ability to lower their growth rates or accumulation. Referred to as allelopathic compounds, several bloom taxa produce spectrums of compounds with these properties. Examples include *Karenia brevis*, the Florida red tide taxon, which produces compounds that kill two competing diatoms and a dinoflagellate (Prince et al., 2008) and retard rotifer grazing on the dinoflagellate (Kubanek et al., 2007). *Alexandrium tamarense* and *A. minutum* produce compounds that had negative effects on the diatom *Thalassiosira weissflogii* and the cryptophyte *Rhodomonas*, as well as a natural community and a ciliate (Fistarol et al., 2004). Cyanobacteria also produce inhibitory compounds, with *Nodularia spumigena*, *Aphanizomenon flos-aquae*, and *Anabaena lemmermannii* all inhibiting these same taxa (Suikkanen et al., 2004). Another dinoflagellate, *Cochlodinium polykrikoides*, inhibits the growth of *Akashiwo sanguinea*, *Gymnodinium instriatum*, and *Chattonella marina* as well as members of the natural phytoplankton community from Long Island estuaries likely through production of reactive oxygen species (Tang and Gobler, 2010); raphidophytes like *C. marina*, *C. antiqua*, *Fibrocapsa japonica*, and *Heterosigma akashiwo* have similar ROS production capacities (Marshall et al., 2005). ROS production can lead to gill damage in fish and death (Kim et al., 1999). The prymnesiophyte *Prymnesium parvum* produces compounds inhibitory to several dinoflagellates (Fistarol et al., 2003). Some macroalgae (*Ulva*, *Corallina*, *Sargassum*, *Gracilaria*, many coral reef cyanobacteria) also produce allelopathic compounds inhibitory to other phytoplankton and macroalgae (Nagle and Paul, 1999; Wang et al., 2007; Tang and Gobler, 2011).

Mechanisms for bloom formation

Planktonic blooms can originate from physical mechanisms (Sellner et al., 2003) like circulation patterns (Langmuir cells, fronts, upwelling), local storms and precipitation, poor mixing, stratification, limited flushing, or periods of low nutrient but abundant light which favors surface accumulations via positive phototactic swimming or buoyancy control of several toxic species. Examples of these physically driven aggregations of cells include frontal concentration of *Gymnodinium* in the Potomac River estuary (Tyler and Heinbokel, 1985) and *Gymnodinium catenatum* in Spanish Rias (Figueiras et al., 1995); blooms of diatoms transitioning to dinoflagellates from wind-induced upwelling of nutrient-rich deepwater in mid-Chesapeake Bay (Malone et al., 1986; Sellner and Brownlee, 1990); rainfall-induced fresher lenses of water overlying salty, dense estuarine water (Loftus et al., 1972); blooms of *Microcystis* in tidal fresh embayments with little exchange (Linkov et al., 2007) or coastal lagoons with limited flushing and long residence times where raphidophyte or brown tide blooms are frequently observed (Gobler et al., 2005; Handy et al., 2005); and dense, water discoloring surface blooms of several dinoflagellates and gas vesicle-rich cyanobacteria in highly stratified and nutrient-poor surface waters (e.g., Sellner and Brownlee, 1990; Suikkanen et al., 2007).

The latter bloom populations have remarkable diel behaviors afforded to them through motility from flagella and gas vesicle buoyancy, respectively. Diurnal surface aggregation is assured through active upward swimming of the dinoflagellates via positive phototaxis (Kamykowski et al., 1998), while in the cyanobacterium, flotation is assured through filling of thick-walled gas vesicles in the cell, causing the cells to rise to the surface. At night, the dinoflagellates may actively swim or passively settle to depth, while the cyanobacterium, now rich in carbohydrate from daylong photosynthesis, sinks due to the carbohydrate ballast that overwhelms the buoyancy of the gas vesicles in the cyanobacterium. While at depth, both groups assimilate nutrients. The dinoflagellates respond to daylight again, actively swimming to the surface. In contrast, nocturnal respiration of the carbohydrate in the cyanobacterium oxidizes the carbohydrate, removing ballast, and buoyancy produced through gas-filled vesicles again returns the cyanobacterium to the surface (e.g., Chu et al., 2007).

Finally, unique life cycles for some of the bloom-forming species ensure continued growth opportunities when light, nutrients, and physical conditions are near optimal (see Fryxell, 1983). Many dinoflagellates (e.g., *Alexandrium*, *Gonyaulax*, *Protoceratium*, *Peridinium*, *Diplopsalis*, *Diplopsalopsis*, *Diplopeltopsis*, and *Scrippsiella*) and raphidophytes (*Heterosigma akashiwo*, *Fibrocapsa japonica*) produce temporary and permanent resting stages (cysts) that on resuspension into the overlying water column can excyst (burst from the cyst wall) to resume the swimming behavior described above. Similarly, some diatoms produce resting spores (e.g., *Chaetoceros*, *Leptocylindrus*, *Bacteriastrum*), enabling the same reemergence and growth possibilities. For the cyanobacteria, some taxa (e.g., *Aphanizomenon*, *Anabaena*) produce resting stages called akinetes and, like cysts and spores, enable regrowth. Others (*M. aeruginosa*) simply settle to the bottom in the fall to overwinter as near-dormant populations on the sediment surface, only to be resuspended in the late spring for reinitiating cell division and accumulation into new blooms.

Summary

Toxic (and nontoxic) blooms of planktonic and benthic single-cell organisms and accumulations of macroalgae are increasingly common in the world's nutrient-rich estuaries, symptomatic of eutrophication attributable to mass land use changes associated with human development. Toxic blooms occur in tidal fresh and brackish portions of the estuary, with a spectrum of toxicities on exposed estuarine fauna as well as bird and mammal consumers: diarrhea, nausea, and respiratory distress are common, while fish and shellfish mortalities can result from both exposure to toxin and low dissolved oxygen concentrations associated with high-biomass blooms of cyanobacteria, dinoflagellates, raphidophytes, and macroalgae. Although many bloom-forming species are favored in nutrient-rich estuaries, there are several physical, physiological, and behavioral mechanisms within estuaries and cells that can overlap to allow growth and concentration of both toxic and nontoxic species to ultimately adversely affect estuarine fauna and larger consumers.

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Cross-references

- Algal Blooms
- Anthropogenic Impacts
- Cyanobacteria
- Density Stratification
- Dissolved Oxygen
- Estuarine Circulation
- Eutrophication
- Flushing Time
- Macroalgae
- Macroalgal Blooms
- Nutrients
- Oxygen Depletion
- Phytoplankton
- Phytoplankton Blooms
- Residence Time

TRACE METALS IN ESTUARIES

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Synonyms

Heavy metals

Definition

A metal is defined as an element that can form a salt by replacing the hydrogen (H^+) in an acid and form a base when combined with the hydroxyl ($-OH^-$) radical.

Description

A few metals rank among the most common elements on earth (Fe, Al, Ca, Mg), but most are less or much less abundant. These can be considered trace metals. Since these comprise most of the elements in the periodic table, we should define those of concern in estuaries as those that are deposited from natural and anthropogenic sources upriver. Typically, those of interest are of anthropogenic origin, since they tend to be more evident downstream of industrial outfalls or mine drainages.

Trace metals can be either essential for, or irrelevant to, living organisms (i.e., nonessential). In regard to living organisms, trace metals are typically defined as those that are essential (Murray et al., 2009). However, in estuaries both essential and nonessential metals should be included in the definition since, in either case, they can be toxic in minute quantities – nanomolar concentrations or less, while those that are essential (micronutrients) are toxic only when they substantially exceed metabolic needs – typically, micromolar concentrations. These metals are found in estuarine sediments, biota, and (to a lesser extent) in the water column. Of these metals, two deserve special attention. While most trace metals are found in living tissues (i.e., they bioaccumulate), Hg and As* are capable of biomagnification. This latter term means that they increase, typically by an order of magnitude, with each trophic level in the food web. This occurs because these metals are turned into organometals (carbon-containing compounds) as a result of metabolic processes. Because of the organic part of the molecule, an organometal can passively cross the lipid bilayer that all cell membranes are made of, rather than requiring an active metabolic process for uptake; they then are retained rather than excreted and are passed on to the next trophic level. Organomercury compounds are particularly toxic. Some organoarsenicals are toxic, while others are relatively innocuous – e.g., the metabolism of inorganic As to trimethylarsine renders it less toxic (Reimer et al., 2010).

Table 1 summarizes the status of trace metals found in estuaries.

Trace Metals in Estuaries, Table 1 Trace metals in estuaries

Metal	Symbol	Essentiality[a]	Capable of biomagnification	Found in sediment	Found in biota
Arsenic[b]	As		X	X	X
Beryllium	Be			X	X
Cadmium	Cd			X	X
Chromium	Cr	X		X	X
Cobalt	Co	X		X	X
Copper	Cu	X		X	X
Lead	Pb			X	X
Manganese	Mn	X		X	X
Mercury	Hg		X	X	X
Molybdenum	Mo	X		X	X
Selenium	Se	X		X	X
Silver	Ag			X	X
Vanadium	V	(X)		X	X
Zinc	Zn	X		X	X

[a] While those metals listed as essential are as such for all eukaryotes and some bacteria, vanadium is known to be essential in relatively few species, including tunicates

[b] Arsenic is included here among "metals," although it may be properly considered a "metalloid"

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TROPHIC DYNAMICS

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Synonyms

Trophic structure; Trophic web

Definition

Trophic dynamics is the basic process of transference of energy from one trophic level to the next in an ecosystem.

Introduction

The basic process of energy transfer in ecosystems occurs through trophic dynamics (Lindeman, 1942). Trophic dynamics in tropical and subtropical estuarine and coastal marine ecosystems are characterized by a broad heterogeneity, which is determined by highly diverse biological communities and complex trophic webs with a high degree of interaction (Manickchand-Heileman et al., 1998). The dynamic nature of trophic web interactions and the variation in structure and function of several coastal systems (including estuaries) are well known.

From the time of naturalists to the present-day ecologists, there has been an enormous interest in understanding energy transfer in ecosystems, and research has increased notably since Lindeman (1942) coined the concept of trophic dynamics. This concept, along with the articles published by Odum, fostered a rapid progress of the use of the laws of thermodynamics in ecological studies.

Estuarine trophic dynamics

Trophic dynamics have been widely studied in estuaries, especially within the ichthyological community, but the role of other biological components in the flow of energy toward different trophic levels has only been suggested. Estuaries are habitats with an exceptional primary productivity that allows them to support a high abundance and biomass of fish; they are essential habitats in the life cycle of several teleosts, functioning as feeding, reproduction, and refuge areas. Fish communities in estuaries are very dynamic due to seasonal changes in abiotic factors: currents, water temperature, pH, freshwater inputs, evaporation, precipitation, and drastic changes in salinity. These changing and extreme conditions result in estuaries being characterized by few dominant fish species that are resident or typical of these areas. These species are classified as anadromous (fish that migrate from salt water to freshwater to reproduce) or catadromous (fish that migrate from freshwater to salt water to reproduce) (Kennish, 1990).

The trophic dynamics of estuaries are determined mainly by the trophic relationships of all biological components. The movement of nutrients between estuaries and marine communities occurs through the biota in a predictable sequence of trophic relations (nutrients). The trophic structure of communities is centered on the concept of the trophic web, which is the result of branched and/or linked trophic chains. It can be defined as the sum of all chains, with organisms grouped in categories or trophic levels, which consist of producers, consumers, and decomposers. These in turn are responsible for energy transfer. Therefore, at the base of the trophic chain are all the autotrophic organisms which fixate carbon through photosynthesis and provide energy to the primary consumers (heterotrophs) and so on through the upper trophic levels. Primary consumers serve as food for secondary consumers (herbivores), and these are a source of food for tertiary consumers (carnivores). Decomposers (saprophytic bacteria and fungi) assimilate dead plant and animal matter, transforming it in organic matter to get energy, while they release mineral nutrients that primary producers (autotrophs) use for growth. At each trophic level, approximately 80–90 % of potential energy is lost as heat, and this is a limiting factor that restricts trophic chain lengths to three to four trophic levels.

Estuarine trophic dynamic processes

The feeding habits of fish are important for several reasons. First, they reveal the trophic relations of the different species and indirectly bring to light aspects of the energy flow in estuarine communities. Second, they reveal the ecological relations between organisms, which can be used to better interpret the general dynamics of estuaries and to make recommendations for the appropriate administration of fisheries resources. The functioning of estuarine and coastal marine ecosystems depends mostly on using an external source of energy, solar radiation. A portion of this incident energy is transformed by photosynthetic processes (primary production) carried out by the phytoplankton (such as diatoms, dinoflagellates, coccolithophorids, cyanophytes, and silicoflagellates) and phytobenthos (red, green, and brown algae) (Lindeman, 1942). These components, along with detritus, constitute the main energy source for organisms in estuaries (Kennish, 1990). Zooplankton is comprised mainly of crustaceans such as copepods, as well as molluscs, coelenterates, and chaetognaths. The phytoplankton is consumed by zooplankton; about 50 % of phytoplankton is not consumed and goes into detritus. The zoobenthos consists of a wide range of organisms, mostly invertebrates such as sponges, crabs, echinoderms, polychaetes, snails, clams, and sea stars, which feed on the phytobenthos as well as on detritus. The zoobenthos is one of the main food sources for fish, which compose the most conspicuous and dominant community in estuaries. The abundance, biomass, and diversity of the ichthyofauna are regulated by the size of the estuary, the season (wet or dry), latitude (tropical and subtropical zones), and changes in abiotic factors (temperature, freshwater input, salinity changes, pH, currents within the estuary, and evaporation). These factors can lead to changes in the fish community structure through migration or immigration of less eurytopic species. However, most of the population is comprised by highly tolerant species that have a clear tendency toward an *r* strategy (Pianka, 1970).

In general, four fish feeding categories can be observed (Claro, 1994):

Herbivores: This trophic category includes fish that feed exclusively on plants (algae). Fish from the Kyphosidae family are very well represented in estuaries.

Detritivores: This group contains species that consume detritus, which consists of vegetal and animal material that has not been consumed and gets deposited on the benthos. The most typical fish family in estuaries is the Mugilidae family.

Benthophagous: Fish that feed on small organisms that inhabit the substrate. Most fish species present in estuaries are found in this category (e.g., Mullidae and Chaetodontidae families).

Carnivores: Consumers of animal prey, they feed mainly on fish and invertebrates. The most representative species belong to the families Lutjanidae, Serranidae, Haemulidae, Carangidae, and Balistidae.

A practical way to analyze the trophic dynamics of estuaries is by performing functional diversity studies that provide a rapid characterization of communities, reducing complex ecosystems into species groups (functional groups) with ecological equivalence, thereby facilitating comparative community studies (Root, 1967; Root, 2001).

Species that comprise a functional group are those that overlap in the highest number of variables in the multidimensional niche, being ecologically equivalent in function (Nagelkerken and Van der Velde, 2004). From the functional viewpoint, biological communities can be represented as a mosaic of functional groups, or as bricks with which communities are built (Krebs, 2003). The presence of multiple species (dominant and subordinates) within each functional group increases functional redundancy and contributes to the resilience of the estuary or ecosystem studied (Hooper et al., 2005).

With a higher number of functionally similar species, each with different responses to environmental factors, there is a higher probability of at least one species surviving possible perturbations (Hooper et al., 2005). If there is no functional redundancy, the loss of a single species could result in the complete loss of a functional group; therefore, at least one species per functional group is essential for ecosystem functioning (Díaz et al., 2005). Currently, due to urban and touristic development, anthropogenic impacts affect negatively coastal areas and especially estuaries, so that immediate and continued evaluations of these effects on the local fauna and flora are extremely important. The study of functional diversity is therefore an effective and precise tool to evaluate these effects.

The most important inputs for the development and evolution of trophodynamic perspectives over the last decades have occurred mainly in the field of aquatic ecology (Lindeman, 1942; Ulanowicz, 2004). The mass balance model ECOPATH with Ecosim has been the most extensively used tool for analyzing trophic webs (Polovina, 1984; Christensen and Pauly, 1992). This model describes quantitatively the energy flows (biomass) within food webs and facilitates an approximation to the structure and function of the ecosystem. Additionally, the model outputs allow the calculation of several ecological indicators of the state of the trophic web, such as ascendancy, surplus, and development potential (Ulanowicz, 1986; Libralato et al., 2006), that can be used to follow up an ecosystem's development through time and to compare the ecological maturity of systems located in different parts of the world.

Summary

Estuarine trophic dynamics is one of the most complex processes of estuarine science. Its complexity lies in the high environmental heterogeneity and biodiversity of these coastal systems, which includes all trophic levels from the smallest autotrophs to the largest predators. The microbial decomposers must be considered as well. Among the most important components in the flow of energy in estuaries are fish communities due to the great diversity of finfish eating habits which has resulted in a large number of direct and indirect trophic interactions with other biotic components present within networks that structure and determine the energy dynamics of these valuable ecosystems.

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Cross-references

Food Web/Trophic Dynamics

TURBIDITY

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Turbidity

Turbidity is a measure of clarity in the water caused by suspended matter, such as clay, silt, fine particulate organic and inorganic matter, soluble colored organic compounds, plankton, and other microscopic microorganisms. Turbidity is an expression of the optical property that causes light to be scattered and absorbed rather than transmitted straight through a water sample. Turbidity measurement is mandatory for regulatory purposes.

In the middle of twentieth century, turbidity was measured using the Jackson candle turbidimeter. The method was visual since the scientist evaluated light scattering using unaided eyes. This method is no longer in use. The modern method is called nephelometric, and it is based on a comparison of the intensity of light scattered by the sample under defined conditions with the intensity of light scattered by a standard reference suspension (usually formazin, a polymer) under the same conditions. Turbidity units depend on the light source (e.g., white, near infrared), its angle in relation to the detector, and number of detectors (i.e., ratio).

Turbidity is expressed in nephelometric turbidity units (NTU) if the light source is white or broadband (400–680 nm) located at 90° detection angle to one detector (Anderson, 2005), nephelometric turbidity ratio units (NTRU) if the light source is at 90° detection angle and measurement is done by multiple detectors with ratio compensation, backscatter units (BU) if the light source is at 30° ± 15° detection angle (backscatter), attenuation units (AU) if the light source is at 180° detection angle (attenuation), and nephelometric turbidity multibeam units (NTMU) if there are multiple light sources and detectors are located at 90° and possibly other angles to each beam.

Sometimes turbidity can be expressed in formazin turbidity units (FTU) comparable to NTU, obtained by absorptiometric method measuring the amount of the transmitted light absorbed by a formazin standard. This method uses near-infrared (780–900 nm) or monochrome light source. Depending on device, units can vary such as FNRU (formazin nephelometric ratio units), FBU (formazin backscatter units), FAU (formazin attenuation units), and FNMU (formazin nephelometric multibeam units) (Anderson, 2005).

Various studies made correlation between turbidity and water quality parameters. The first study to use turbidity for computation of suspended loads was presented by Emmet (1975) and Truhlar (1976). Effler and Johnson (1987) showed correlation between turbidity and calcium carbonate precipitation in lakes. LeChevallier and Norton (1992) showed that the removal of particles greater than five microns and turbidity were useful predictors of Giardia and Cryptosporidium removal. Strunk (1992) identified correlation between turbidity and sewage treatment plants and road discharge.

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TYCHOPLANKTON

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Synonyms

“Tychopelagic” forms, in reference to diatoms (Hendey, 1964)

Definition

Hendey’s (1964) description of marine tychopelagic diatoms specified that these forms spend a major portion of their life cycle attached to fixed substrata but become planktonic after physical processes suspend them or tear them loose from substrate. Hutchinson (1967), examining freshwater forms, presented a broader definition of tychoplankton, which included pseudoplankton (only occasionally suspended in the water) and meroplankton (spending a part of their life cycle in the plankton). Kuhn et al. (1981) defined tychoplanktonic species as occupying benthic niches but under certain conditions proliferating in the plankton. The utility of distinguishing tychoplankton includes studies of past conditions (paleoecology).

Taxonomy

Most published references to tychoplankton refer to diatoms that are typically found on the bottom or other solid substrate and that enter the water column when physical forces (waves, tidal currents) suspend them. Owing to the difficulty of distinguishing tychoplankton from obligately benthic or planktonic forms except by identification to species and counting, estimates of tychoplankton biomass and production are challenging, although tychoplankton are likely ubiquitous in estuarine ecosystems. Inclusion of meroplankton, species regularly spending part of their life cycle in the plankton, with the tychoplankton is not a common usage in the literature on estuarine ecology.

Example species

Vos and DeWolf (1993) identified the diatoms *Cymatosira belgica* and *Raphoneis minutissima* as tychoplankton species. Espinosa et al. (2006) identified the diatoms *Paralia sulcata*, *Staurosira construens*, and *Staurosirella pinnata* as tychoplankton.

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Cross-references

Phytoplankton
Wave-Driven Sediment Resuspension

U

UPLIFTED COASTS

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Synonyms

Coseismic uplift; Crustal uplift; Emergence; Postglacial uplift; Raised beaches

Definition

An *uplifted coast* is one that undergoes continual or stepwise uplift (Mörner, 1980).

Emergence implies that the shoreline is displaced seaward and that land emerges from the sea. *Postglacial uplift* denotes the process of *glacial isostasy* (Jamieson, 1882; De Geer, 1888/90); in former glaciated areas (e.g., Fennoscandia, Scotland, northern North America, Patagonia), land rose when the load of ice disappeared after the last ice age peaking at about 20,000 BP. This gives rise to sequences of *raised beaches* expressed in time/elevation *uplift curves* (e.g., Hillair-Marcell and Fairbridge, 1978; Mörner, 1979). Active plate margins of subduction (e.g., eastern South America, southern Italy, Japan, New Guinea) generally experience *crustal uplift*, classifying these areas as *uplifted coasts*. *Coseismic uplift* refers to the state when uplift occurs in steps triggered by earthquakes. This is the case in many places in Japan (Ota, 1986) as well as in New Zealand (e.g., the classical Turakirae shorelines described by Wellman, 1967).

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V

VULNERABILITY INDEX

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Synonyms

Contingency index; Susceptibility index

Definition

A vulnerability index (VI) is a tool to magnify or measure the effects of any potential hazard (natural or human-induced environmental, social, or economic hazard/risk) where a population or region is exposed. The first formal VI was proposed by the Maltese Ambassador to the United Nations in 1990. The basics of a VI were described by researcher Lino Briguglio of the University of Malta. The index was formalized by the ratio between the quantitative components to the relative importance and the cumulative score of the weighted values (Sullivan, 2008).

Description

A VI can be generated to define vulnerability of coastal areas or ecosystems (e.g., estuaries, lagoons, coastal wetlands) to sea-level rise, climate change, floods, oil spills, erosion, etc. This phenomenon is called the coastal vulnerability index (CVI). Indicators selected from ecological, social, or economic components can be used to understand vulnerability of coastal systems to a certain exposure. Guannel and Verutes (2012) used a model in order to achieve qualitative assessment of vulnerability (varying from very low, rocky cliffs to very high deltas) based on the mixture of relative and absolute ranking of variables (e.g., geomorphology, relief, natural habitats, sea-level change, wind exposure, surge potential).

In order to apply a VI to a specific coastal area, the distribution of coastal environments (e.g., sandy beaches, dunes, lagoons, coastal wetlands, estuaries, rocky cliffs) must first be mapped and classified in terms of potential vulnerability to a specific hazard (Doukakis, 2005). A CVI is an effective tool to emphasize areas where vulnerability to any hazard may be the greatest. Once each section of coastline is assigned a vulnerability value for each specific data variable, a CVI is calculated as the square root of the product of the ranked variables divided by the total number of variables (ASIS, 2004):

$$\mathrm{CVI} = \sqrt{\frac{a+b+c+d+e+f}{6}} \tag{1}$$

where:
a = geomorphology
b = shoreline erosion/accretion rate
c = coastal slope
d = relative sea-level rise rate
e = mean wave height
f = mean tide range

All factors were scaled, so that positive values indicate higher levels of vulnerability, while negative values decreased vulnerability. Physical VI scores range from 21.857 to 2.490, with a median value of 0.546 (Boruff et. al., 2005).

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Cross-references

Climate Change
Oil Pollution

W

WASHOVER FANS

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Synonyms

Event deposits; Overwash deposits; Storm deposits; Washovers; Wave deltas

Definition

Washover fans, termed by Price (1947), are plume- to lobate-shaped backshore sandy deposits that form as the result of storm overwash. They can range in thicknesses from millimeters to meters and widths from several meters to kilometers and typically form behind or adjacent to sandy barrier islands, strandplains, and shoreface deposits. Fans consist of landward-dipping sand sheets that typically interfinger with finer-grained back-barrier sediments. Washover fans are found in modern and ancient sedimentary settings.

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Cross-references

Overwash
Washovers

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WASHOVERS

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Synonyms

Event deposits; Overwash deposits; Storm deposits; Washover fans; Wave deltas

Definition

Washovers, termed by Lobeck (1939), are resultant backshore sandy deposits formed by the process of overwash during storms.

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Cross-references

Overwash
Washover Fans

WATER CLARITY

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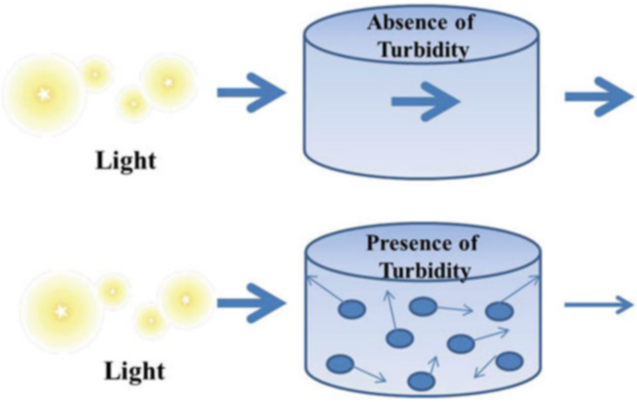
Definition

Water clarity is defined as the depth to which light penetrates water and refers to the transparency or clearness



Water Clarity, Figure 1 Schematic of light attention through the water column in the absence and presence of suspended solids (Schematic by Melanie D. Harrison).

Water Clarity, Figure 2 Image of poor water clarity in Minebank Run in Baltimore, Maryland, due to the presence of suspended sediment caused by urban storm runoff (Image courtesy of Melanie D Harrison).

of water. How far down light penetrates through water depends on many factors that change the light-attenuating properties of the water column. Water clarity is reduced by the presence of suspended and colloidal materials such as clay, silt, finely divided organic and inorganic matter, and plankton or other microscopic organisms.

Description

Suspended materials reduce water clarity by absorbing and scattering light. Turbidity is often used as a general term to describe the lack of transparency or "cloudiness" of water due to suspended or colloidal materials (Figure 1). An increase in turbidity results in a corresponding decrease in water clarity.

Water Clarity, Figure 3 The microprocessor-based waterproof TN100 m is an easy-to-use turbidimeter commonly used to measure turbidity (Image courtesy of Cole-Parmer Corporation and Eutech, Inc. http://www.eutechinst.com/pdt-para-turbidity-tn100.html).

Source: High turbidity may occur from an increase in phytoplankton (algae) or an increase in sediments. This may be in response to nutrient inputs and erosion due to changes in land use, point source and nonpoint source discharges, or agricultural practices. Human impacts such as urbanization, forestry, agriculture, industries that discharge into waterways, and mining are all likely to increase turbidity of nearby waters, particularly when these impacts involve large quantities of storm water and surface runoff (Figure 2).

Effects: Turbidity affects organisms that are directly dependent on light, such as aquatic plants, because it limits their ability to carry out photosynthesis. Excessively high concentrations of suspended particles can have the following effects: (1) limiting light available for photosynthesis; (2) lowering dissolved oxygen concentrations; (3) damaging gills and suffocating fish and oysters; and (4) interfering with filter feeding by benthic organisms. Other organisms that depend on the plants for food and oxygen will also be impacted.

Methods for Measuring of Water Clarity: Water clarity can be measured with a Secchi disk and is the simplest, most inexpensive, and one of the most commonly used relative indicators of turbidity since clarity decreases as turbidity increases. A second commonly used instrument to measure water clarity is an electronic turbidimeter, recorded in formazin turbidity units (FTU) or nephelometric turbidity units (NTU) (Figure 3). This instrument measures the amount of light scattering that occurs within a given water sample by shining a bright light on one side of the sample and measuring the amount redirected to the detector located 90 ° to the incident light (direction of the incoming light). Scattering from the water sample is measured relative to the amount of light scattered by a reference solution (a solution that will cause a known amount of light scattering; Davies-Colley, 1990; Davies-Colley and Smith 2001). At larger scales, remote sensing-based methods are well suited for monitoring clarity in water bodies such as inland lakes, rivers, and estuaries (Woodruff et al., 1999; Kilham et al. 2012). Robust water clarity estimates can be derived using MODIS (or Moderate Resolution Imaging Spectroradiometer), a key and commonly used instrument combined with field-collected Secchi disk transparency data to estimate water clarity. The use of remote sensing data combined with transparency data is an effective tool for measuring changes in water clarity throughout the growing season (Knight and Voth, 2012).

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Cross-references

Secchi Disk
Turbidity

WATER QUALITY

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Definition

Water quality is broadly defined as the physical, chemical, and biological composition of water as related to its intended use for such purposes as drinking, recreation, health of ecosystems, and safety of human health (Veatch and Humphrys, 1966; Richard and McQuisten, 1968; APHA, 1969). The term can be applied to a single characteristic of water or to a group of characteristics combined into a water quality index.

Impacts to water quality

Anthropogenic impacts coupled to urbanization as well as agricultural, commercial, and industrial uses greatly affect the water quality of lakes, streams, rivers, inland bays, and coastal estuaries. These changes in land use increase pollutant loads (e.g., pesticides, nutrients, sediments, trace metals, and volatile organic carbons) and have adverse effects on water quality in the environment (Gilliom et al., 2006; Horowtiz and Stephens, 2008; Spargue et al., 2009; Dubrovsky et al., 2010). Coastal estuaries experience harmful impacts to water quality (e.g., eutrophication, low dissolved oxygen, harmful algae blooms, and sediment contamination) as a result of land use drivers of change (USEPA, 1999) (Figure 1).

Water quality standards

In surface waters, water quality standards are developed to assess water quality as it relates to health of ecosystem, safety of human contact, and drinking water.

Water Quality, Figure 1 Satellite imagery of the Chesapeake Bay; the largest estuary in the United States (Image courtesy of Flickr, tag eutrophication and hypoxia).

A water quality standard is defined as the designated beneficial use(s) of a water segment and the water quality criteria necessary to support those uses (CWA, 1972). Water quality standards establish the environmental baselines used for measuring the health of aquatic ecosystems.

Measuring water quality

The parameters for water quality are determined by the intended use. Water quality is measured by several factors, such as the concentration of dissolved oxygen, bacteria levels, nutrients concentrations (commonly nitrogen and phosphorus), amount of salt (or salinity), or the amount of material suspended in the water (turbidity). In some bodies of water, the microscopic algae and quantities of pesticides, herbicides, heavy metals, and other contaminants may also be measured to determine water quality. Parameters commonly measured in water bodies include temperature, pH, turbidity, Secchi disk depth, dissolved oxygen, conductivity, and oxygen reduction potential (ORP). Biological metrics have been developed in many places, and one widely used measure is the presence and abundance of members of the insects. Biological indexes vary from region to region; however, the greater the number of taxa from these orders, the better the water quality.

Water quality models

Various water quality models have been used to simulate the fate of pollutants and the state of selected water quality variables in water bodies. They incorporate a variety of physical, chemical, and biological processes that control the transport and transformation of these variables. For example, the Water Quality Analysis Simulation Program (WASP7) is one of the most widely used water quality models in the United States and throughout the world (Di Toro et al., 1983; Connolly and Winfield, 1984; Ambrose et al., 1988). Users are able to interpret and predict water quality responses to natural phenomena and man-made pollution for various pollution management decisions. AQUATOX is a simulation model for aquatic systems and predicts the fate of various pollutants, such as nutrients and organic chemicals, and their effects on ecosystems, including fish, invertebrates, and aquatic plants.

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Algal Blooms
Anoxia, Hypoxia, and Dead Zones
Biochemical Oxygen Demand
Clean Water Act
Eutrophication
Nonpoint Source Pollution
Nutrients
Oil Pollution
Oxygen Depletion
Pathogens
Water Quality

WATER RESOURCES AND CLIMATE CHANGE

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Synonyms

Hydrometeorology

Definitions

Hydrology is the study of space-time characteristics of the quantity and quality of the water on earth. It includes the occurrence, movement, distribution, circulation, storage, exploration, development, and management of water in space, time, and frequency domains. Hydrology is partitioned into surface water hydrology, groundwater hydrology, snow and glacial hydrology, and atmospheric hydrology or hydrometeorology.

The hydrologic cycle is the continuous movement of water between the atmosphere, biosphere, pedosphere, lithosphere, and hydrosphere. It is the core of hydrologic research.

Water resources refer to the water stored in any form on the earth surface and in the atmosphere, biosphere, pedosphere, and lithosphere. More specifically, water resources are the waters feasible for human use (their quantity and quality) that can be renewed within a reasonable time interval.

Hydrologic cycle

Evaporation is the only hydrologic component connecting water balance and energy balance and is the key component in understanding the hydrologic cycle (Xu and Singh, 2005). Xu et al. (2006) summarized the terms describing evaporation in the literature: (1) free water evaporation, ET_o, represents the amount of evaporation from open/free water surface; (2) actual evapotranspiration, ET_a, describes all the processes by which liquid water at or near the land surface becomes atmospheric water vapor under natural conditions; and (3) potential evapotranspiration, introduced in the late 1940s and 1950s by Penman (1948, 1956), is defined as "the amount of water transpired in a given time by a short green crop, completely shading the ground, of uniform height and with adequate water status in the soil profile." Reference evapotranspiration (ET_{ref}) (Allen et al., 1998) is defined as "the rate of evapotranspiration from a hypothetical reference crop with an assumed crop height of 0.12 m, a fixed surface resistance of 70 s m^{-1} and an albedo of 0.23, closely resembling the evapotranspiration from an extensive surface of green grass of uniform height, actively growing, well-watered, and completely shading the ground." Evapotranspiration is perhaps the most difficult of all the hydrologic components in the hydrologic cycle due to complex interactions among the components of the land-plant-atmosphere system (Xu and Singh, 2005; Zhang et al., 2011b).

It is widely expected that the increasing average global temperature will increase evaporation of terrestrial water bodies. However, many studies have found decreased pan evaporation and ET_{ref} (e.g., Roderick and Farquhar, 2002), which is usually called the pan evaporation paradox. Global warming has the potential to alter the hydrologic cycle and therefore cause uneven distribution of water resources. Hydrologists and meteorologists suggest that an increase in surface temperature can result in higher evaporation rates and enable the atmosphere to transport higher amounts of water vapor which, in turn, leads to an accelerated hydrologic cycle (e.g., Menzel and Bürger, 2002). Warmer temperatures increase the capacity of the atmosphere to hold water vapor causing greater probability of high-intensity rainstorms, hence triggering flood and drought conditions. Global warming therefore may result in an increase in intensity of extreme events (WMO, 2003) and the uncertainty of water supply at regional and global scales.

Climate changes and weather extremes

Climate change reflects changes in the statistical distributions of weather components at various time scales ranging from seconds to millions of years. The weather components include precipitation, temperature, evaporation, relative humidity, and so forth. The causes behind climate change can be oceanic and biotic processes, solar radiation changes, plate tectonics and volcanic eruptions, and also human-induced alterations of natural processes. Climate change is sometimes referred to as global warming which can cause different hydrologic responses at different spatial and temporal scales.

A trend of decreasing rainfall has been documented in the Mediterranean area (Karl, 1998), and significant decreasing extreme rainfall has also been observed in Western Australia (Haylock and Nicholls, 2000). However, some researchers have found increasing extreme precipitation in the United States (e.g., Kunkel, 2003). Others have reported no trend in extreme rainfall in Canada (Zhang et al., 2001). Zhang et al. (2011a) also indicated changing characteristics of precipitation across China. Generally, a wet tendency has been noted in south China, a slight wet tendency in northwest China, and a dry tendency in north China. Further, increasing precipitation intensity has been witnessed mainly in the lower Yangtze River and the Pearl River basins.

Analyzing wet periods and the associated precipitation intensity over Europe, Zolina et al. (2010) indicated that longer wet periods and higher intensities should have a significant impact on the terrestrial hydrologic cycle, including subsurface hydrodynamics, surface runoff, and

European flooding. However, different hydrologic responses to precipitation changes have been identified in China, where altered precipitation changes are mirrored by shortening consecutive precipitation episodes and shifts of seasonal precipitation changes (Zhang et al., 2011a). Uneven spatiotemporal distribution of precipitation changes and also different hydrologic responses to climate change in different regions of the globe result in different types of variability and availability of water resources over the earth.

Flood and drought hazards may be the most serious natural disasters impacting human society. Economic losses due to floods and droughts have been increasing with population growth and development and associated poor land use in the floodplains of great rivers (Changnon, 1985). A flood is an overflow of water that inundates land areas, and a flood disaster refers to a flood event that causes socioeconomic loss or mortality. Droughts are long-term phenomena affecting large regions inflicting significant damages on both human lives and socioeconomic conditions. A drought event is a period characterized by insufficient precipitation, soil moisture, and water resources for supporting the socioeconomic activities of a region. However, there is not a universal definition of drought due to the wide variety of sectors affected by a drought, its diverse geographical and temporal distribution, and the demand placed on water supply by human-use systems. The most well-known and widely used classification of droughts is the classification initially proposed by Dracup et al. (1980) and subsequently integrated by Wilhite and Glantz (1985). The American Meteorological Society (2004) adopted this drought classification system. Based on the nature of the water deficit, four types of droughts are defined: (1) meteorological drought, (2) hydrologic drought, (3) agricultural drought, and (4) socioeconomic drought. This work focuses on meteorological drought which is defined as a deficit of precipitation over a region within a certain time interval.

Water resource management

Water resource management focuses on planning, allocating, developing, distributing, and managing the optimum use of water resources, and the final objective is to make good use of limited water resources and to optimally satisfy water demands of various sectors. Climate change is expected to alter the spatiotemporal distribution of precipitation and evapotranspiration patterns (Tsanis et al., 2011). These alterations, in turn, cause variability in the availability of water resources, river discharge, and seasonal water supply (Arnell et al., 2011; Koutroulis et al., 2013). The demand for freshwater is rising, but a variety of factors, including population growth, water pollution, economic progress, land use change, and climate change, render its availability uncertain in the future (Davies and Simonovic, 2011). Awareness of greater water shortages has aroused an increasing interest in the planning of water resources (Alcamo et al., 2003), with the aim to develop and implement appropriate water resource infrastructure and management strategies (Davies and Simonovic, 2011).

Global warming, as a result of human-induced emission of greenhouse gases, has the potential to significantly affect the hydrologic cycle (IPCC, 2007), which is intensifying with the extra precipitation being unequally distributed around the globe (Arnell, 1999). Uneven spatial distribution of precipitation and major shifts in the timing of wet and dry seasons further enhance the uncertainty of water supply and the magnitude of water shortages in both space and time. Water resources are vital for the environment, economy, and society. Therefore, changes in the water resource base have the potential to significantly impact environmental quality, economic development, and social welfare. How to allocate, use, and save water resources for sustainable development of human society and conservation of the environment is a critical element which human beings are facing in the changing environment.

In 1997, the United Nations published a Comprehensive Review of the Freshwater Resources of the World (WMO, 1997). The assessment included four components: (1) the collation of up-to-date national-level data on water resources and their use, (2) the development of projections of future use (to 2025 and 2050), (3) the description of present and future pressures, and (4) the assessment of strategies and options for the sustainable development of world water resources. It highlighted the effects of increasing population and economic development on water resource availability. It also estimated that approximately one-third of the world's population currently lives in countries experiencing moderate to high water stress and predicted that by 2025 as much as two-thirds of the larger world population could be under stress conditions simply due to the increase in population and water use. Arnell (1999) investigated the effects of climate change on water resources globally, noting that the average annual runoff will increase in high latitudes, in equatorial Africa and Asia, and Southeast Asia, but will decrease in midlatitudes and most subtropical regions. In addition, the rise in temperature associated with climate change will lead to a general reduction in the proportion of precipitation falling as snow and a consequent reduction in many areas in the duration of snow cover. This has implications for the timing of streamflow in such regions with a shift from spring snow melt to winter runoff.

Summary

Water resources are the key to the water security, the food security, and hence the sustainable development of human society. Altered hydrologic cycles due to human-induced global warming have the great potential of modifying spatiotemporal distribution of water resources. In addition, climate and hydrologic extremes such as floods, droughts, typhoons, rainstorms, and so on are occurring at greater frequency and intensity which will cause increasing loss of human life, property, and the environment. It is important to increase the understanding of climate change and the influence of human activities on climate change, which is of great relevance to water resource management and also to the planning of agricultural activities. Multidisciplinary research is necessary since climate change and associated hydrologic variations are the result of many factors that require further study.

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Cross-references

Climate Change
Water Quality

WATERSHED

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Synonyms

Drainage basin

Definition

A watershed is an area of land that drains into a given stream, river, lake, or wetland. The shape and size of watersheds vary. They have no county, state, and national boundaries (URL, 1).

Description

An area of land that drains into a given stream, river, lake, or wetland is called a watershed. Surface water run-off along with discharged groundwater merge with streams during their transport and eventually intertwine at the intersection of their valleys. Great rivers are formed by the intersection of large watersheds which then flow into estuaries and oceans (Otto et al., 2004).

Watersheds are geohydrologic basins defined in terms of the boundaries of their water catchment area. They are almost always drained by river courses. All land, water, and biotic resources within their hydrologic boundaries are included in the watersheds which behave as ecological entities. Similar to islands, the boundaries of a watershed can also be marked, and its components can be interlinked. While water is the common factor, other aspects of individiual watersheds such as geology and vegetation modulate the quality and quantity of the water flow. Each watershed can be divided into interconnected tributaries. The changes that occur in some cases may not be significant; however, in other cases, drastic changes may occur (Clark, 1996).

Rivers are linked in a watershed (Otto et al., 2004). The land drained by a river and its creeks is part of a watershed (Smardon et al., 1995). A channel network resembling the branches of a tree forms when smaller streams come together in a watershed generating a drainage network. A drainage basin is where water is fed to the drainage network, and the main channel size, along with its flow rate, increases in proportion with the size of the drainage basin (Marsh, 2010).

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WAVE-DRIVEN SEDIMENT RESUSPENSION

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Definition

Sediment resuspension occurs when sediment is raised from the bed into the water column by fluid forces, including those exerted by waves. This entry discusses sediment resuspension by waves (see *Sediment Transport*).

Description

Field and modelling studies have firmly established that gravity waves in estuaries are capable of resuspending bed sediments (e.g., Ward et al. (1984), Schoellhamer (1995), de Jonge and van Beusekom (1995), Green et al. (1997), Carniello et al. (2005), Talke and Stacey (2008)), and this includes very small waves, less than 20 cm high (Anderson, 1972; Dyer et al., 2000; Uncles and Stephens, 2010; Green, 2011). Unlike tidal currents, which are periodic, waves occur episodically, and the corresponding episodic sediment resuspension adds to or even masks any periodic resuspension driven by tidal currents. For example, Green et al. (1997) described a New Zealand mesotidal intertidal flat where resuspension was entirely due to episodically occurring waves; Christie et al. (1999) reported an order-of-magnitude increase in suspended-sediment concentration (SSC) during storms compared to during fair weather on an intertidal flat in the Humber Estuary (UK); Ralston and Stacey (2007) described how SSC on an intertidal flat in San Francisco Bay drops rapidly as winds and waves abate.

Sediment resuspension can be meaningfully divided between that caused by waves and that caused by currents (see “*Tidal Hydrodynamics*”). The latter includes currents driven by tides, gravitational circulation, and wind. Waves may be generated locally by wind acting on the fetch inside the estuary basin, or they may have been formed in the ocean and propagated into the estuary (de Lange and Healy, 1990; Wright et al., 1992; Talke and Stacey, 2003). Waves and currents vary on different timescales: whereas periods of locally generated waves are 0.5–5 s and ocean-wave periods may exceed 20 s, currents vary on timescales of tens of minutes to hours. Hence, for the same free-stream current speed, the wave benthic boundary layer is thinner, the wave-induced bed shear stress is correspondingly larger, and the wave is therefore more effective at resuspending bed sediments. Experimental observations confirm this. For example, Schoellhamer (1995) showed that waves generating bottom orbital speeds of 15–20 cm/s dominated sediment resuspension at two depths in Old Tampa Bay, Florida, even though mean currents also attained speeds of 15 cm/s. In addition, de Jonge and van Beusekom (1995) found that, in the Ems-Dollard Estuary (northwest Europe), a doubling of wind speed caused a much greater increase in SSC than a doubling in tidal-current speed, leading them to conclude that the "effect of wind on the suspended matter concentration appears to be predominant." Christiansen et al. (2006) showed that SSC was controlled more by the wave-induced bed shear stress than by the current-induced stress on a microtidal intertidal flat in the Danish Wadden Sea.

Statistical models have been developed to explain and predict wave resuspension, and many of these have used various formulations of the wind speed as the independent variable in regression models. For example, de Jonge and van Beusekom (1995) expressed SSC as a linear function of the "effective wind speed," which they defined as the wind speed averaged over three high-water periods preceding sampling, and Dyer et al. (2000) found a good correlation between SSC and wind speed above a certain threshold. Such models are not readily applicable to sites other than the site for which they were developed. Other researchers have used wave parameters in regression models, which make them more generally applicable. For example, both Anderson (1980) and Janssen-Stelder (2000) explained SSC in terms of wave height, and Schoellhamer (1995) correlated SSC against wave height, wave-orbital speed at the seabed and wave-induced bed shear stress. Christiansen et al. (2006) developed regressions of SSC on the wave-induced bed shear stress. Green and MacDonald (2001) showed that the onset of resuspension of fine-grained noncohesive sands was predictable in terms of the Shields parameter θ'_{w}, which is the nondimensional wave-induced skin friction:

$$\theta'_{\mathrm{w}} = \tau'_{\mathrm{w}}/[(\rho_{\mathrm{s}} - \rho)gD] \tag{1}$$

where ρ_{s} is sediment density, ρ is water density, g is acceleration due to gravity, and D is bed-sediment grain size. τ'_{w} is the skin-friction component of the total wave-induced bed shear stress τ_{w} and

$$\tau'_{\mathrm{w}} = \frac{1}{2}\rho f'_{\mathrm{w}} U_{\mathrm{b}}^2 \tag{2}$$

where U_{b} is the wave-orbital speed at the bed and f'_{w} is the skin-friction wave friction factor, which is a function of the grain roughness of the seabed, $k_{\mathrm{g}} = 2.5D/30$:

$$f'_{\mathrm{w}} = \exp\left[5.213\left(k_{\mathrm{g}}/A_{\mathrm{b}}\right)^{0.194} - 5.977\right] \tag{3}$$

where $A_{\mathrm{b}} = U_{\mathrm{b}}\,T/2\pi$ is the wave-orbital semi-excursion at the bed and T is the wave period. Green and MacDonald (2001) also related SSC to θ'_{w}:

$$C_{\mathrm{ref}} = 0.005\rho_{\mathrm{s}}\theta'^{3}_{\mathrm{w}} \tag{4}$$

where C_{ref} is the time-averaged suspended-sediment concentration "very close to" the bed (in this case, within 1 cm) and $\theta'_{\mathrm{w}} > 0.14$.

Wave height H and period T combine with the local water depth h to determine the wave-orbital speed at the bed U_{b}. The bed orbital speed scales directly with wave height and, under all but shallow-water waves, bed orbital speed varies strongly with the inverse of the depth. Even in quite shallow water, orbital motions may not penetrate down from the sea surface to the seabed. For instance, a 1.5-s period wave with a wavelength of 3.5 m is a deep-water wave in water depths greater than just 1.75 m, and the wave-orbital speed at the bed in that case will be virtually zero. Hence, waves are more effective at resuspension on intertidal flats than in channels, because the former are shallower, and temporal (rising and falling tide) and spatial (channels, sloping intertidal flats) variations in water depth translate into temporal and spatial variations in wave resuspension.

Wave resuspension tends to switch off at some point in a rising tide and switch back on again in the falling tide as the part of the water column in which wave-orbital motions strong enough to resuspend sediments is raised and lowered by the tide. Janssen-Stelder (2000) described a characteristic temporal variation in SSC under waves, with concentration peaks occurring at the beginning of the flooding period and at the end of the ebbing period, when wave-orbital motions at the bed were strongest. Green and Coco (2007) observed SSC under waves decreasing with increasing water depth. SSC was a maximum around low tide, when waves overhead were smaller, compared to high tide because of a reduced fetch, but depth attenuation of orbital motions was less because it was shallower.

The so-called wave-driven "turbid fringe," which occupies shallow water around the edges of the estuary and which sweeps up and down the intertidal flat with the tide, is a readily noticeable feature of estuaries during windy conditions. Within the turbid fringe, SSC is maintained by waves that are capable of penetrating to the bed. Uncles and Stephens (2010) noted small breaking waves and associated resuspension in the very shallow leading and trailing edges of advancing and retreating waters of the Tavy Estuary (Tamar River, UK). The waves combined with peak flood and peak ebb tidal currents that occur shortly after immersion and shortly before drying to enhance the turbid fringe. Christie et al. (1999) also noted that a wave-induced turbid fringe could add to high concentrations that occur at the leading edge of the tide. Ruhl et al. (2001) mapped zones of high SSC in shallow sub-embayments of San Francisco Bay associated with wind waves, finding that wind had more effect on SSC during spring tides, which they tentatively attributed to more energetic spring tidal currents keeping sediments scoured from the bed by waves in suspension for longer periods.

Le Hir et al. (2000) developed a model of wave attenuation by frictional dissipation across an intertidal flat that shows how temporal and spatial patterns in stress – and therefore resuspension – arise. For waves impinging on an intertidal flat with uniform slope β, the wave-induced bed shear stress τ_w at any stage of the tide was shown to be maximum at a mid-depth on the flat given by $h_s = H/(15\pi\beta/4f_w)$, where h_s is the depth at the outer edge of the zone of wave saturation and f_w is the wave friction factor. At that depth h_s,

$$\tau_w = \frac{\rho g f_w}{8} \frac{15\pi}{4} \frac{\beta}{f_w} H \tag{5}$$

Here, f_w may be estimated by (3) but with the total bed roughness used in place of k_g, which represents any bedforms that may be present. Inside the wave-saturation zone (i.e., $h < h_s$), wave height is, by definition, proportional to water depth, and τ_w also decreases in proportion to the water depth:

$$\tau_w = \left(\frac{\rho g f_w}{8}\right) \left(\frac{15\pi\beta}{4f_w}\right)^2 h \tag{6}$$

Outside the wave-saturation zone ($h > h_s$), τ_w is inversely proportional to the water depth:

$$\tau_w = \left(\frac{\rho g f_w}{8}\right) \left(\frac{H^2}{h}\right) \tag{7}$$

The variation in τ_w over the tidal cycle at any point on the intertidal flat can be deduced from the cross-flat distribution of stress. Towards the top of the flat where h is always less than h_s, τ_w always varies in phase with the water depth, and the tidal-cycle-maximum τ_w therefore occurs once per tidal cycle, at high tide. At any other location, τ_w is proportional to h early in the flood and late in the ebb when $h < h_s$, but between early flood and late ebb when $h > h_s$ τ_w varies inversely with h. Hence, τ_w peaks twice per tidal cycle, once between low tide and high tide on the flooding tide and a second time between high tide and low tide on the ebbing tide.

Where waves and currents are both present, they interact nonlinearly in the bottom boundary layer such that both the wave component of the total stress and the current component of the total stress are enhanced over their respective pure-flow values (e.g., Soulsby et al., 1993). Bricker et al. (2005) confirmed the mechanism by showing that, at a shallow site in San Francisco Bay, the steady-flow drag coefficient was as much as an order of magnitude greater when waves were present compared to when they were absent. Verney et al. (2007) found that the combined wave–current bed shear stress τ_{wc} in the presence of waves was up to one order of magnitude higher than the bed shear stress attributable to the tidal current in the absence of waves on intertidal flats at the mouth of the macrotidal Seine River Estuary and that τ_{wc} determined both the erosion and deposition of sediment on the cohesive fine-sediment bed. Talke and Stacey (2003) described the wave climate on an intertidal flat in San Francisco Bay (USA) that comprised both locally generated waves and ocean waves that propagated into the bay from the adjacent ocean. They noted that, although the ocean waves contributed only around 10 % of the kinetic energy over a two-week spring–neap cycle, bed stress was greatly enhanced by interaction between the ocean swell and the tidal current. When the bed shear stress in the absence of waves was close to critical for sediment motion, even the small additional stress by ocean swell can result in resuspension. Nowacki and Ogston (2012) demonstrated that the minimum SSC was set by τ_{wc} for a site in mesotidal Willapa Bay (Washington, USA).

Pressure fluctuations under waves can cause a cohesive-sediment bed to fluidize, in which state it becomes fluid-supported instead of particle-supported, leading to the formation of a confined, high-concentration fluid–mud layer close to the bed (McAnally et al., 2007). Sanford (1994) showed that locally generated wind waves in the upper Chesapeake Bay can increase bed erodibility considerably in the short term, which they tentatively attributed to a change in consolidation state due to fluidization of the upper bed sediments by the waves. Wolanski and Spagnol (2003) concluded that waves fluidized subtidal muddy sand beds in King Sound (Australia), even though fluid mud was not observed at the site. The reason, they suggested, was that any fluid mud formed is quickly transported away by tidal currents. Lambrechts et al. (2010) have proposed that the cohesive-sediment erosion rate be partitioned into a component that is due to dislodgement by the bed shear stress of aggregates of individual particles and another component that is due to the buildup of wave-induced pore pressure, where the latter component is proportional to the third power of the significant wave height.

Summary

Even quite small waves, either alone or combined with currents, resuspend sediments in estuaries. There are distinctive temporal (over tidal cycles) and spatial (across intertidal flats) patterns of wave-induced bed shear stress and associated resuspension. The turbid fringe, which is a readily noticeable feature of estuaries, is maintained by waves that are capable of penetrating to the bed. Waves may fluidize subtidal sediments causing an increase in bed erodibility. For further information, Green and Coco (2014) provide a more detailed review of wave-driven sediment resuspension and transport in estuaries.

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Cross-references

Sediment Transport
Tidal Hydrodynamics

WELL-MIXED ESTUARY

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Definition

A well-mixed estuary is a system in which the water column is completely mixed, making the estuary vertically homogeneous.

Description

Mixing is the process whereby a water parcel or water mass is diluted by, or redistributed within, other water masses. When the tidal range is very large (and accompanied by strong tidal mixing), there is sufficient energy to mix the different layers in the water column. Turbulence completely breaks down the vertical salinity stratification so that the water column becomes vertically homogeneous (Dyer, 1997).

Hansen and Rattray (1966) generated a "stratification diagram" to classify the nature of mixing in estuaries and the sensitivity of the stratification to changing conditions (see also Prandle, 2009). The physical conditions in this type of estuary can cause lateral variations in salinity and current velocity with a well-developed horizontal circulation, or, if the lateral mixing is also intense, the estuary can become sectionally homogeneous. Pritchard (1955) and Cameron and Pritchard (1963) subdivided well-mixed estuaries into laterally inhomogeneous systems (when the estuary is sufficiently wide, the Coriolis and centrifugal forces cause a horizontal separation of the flow) and sectionally homogeneous systems (when the width is smaller and lateral shear may be sufficiently intense to create laterally homogeneous conditions).

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Cross-references

Estuarine Circulation

WETLANDS

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Synonyms

Bog; Fen; Mangal; Marsh; Mire; Morass; Peatland; Quag; Swamp

Definition

Wetlands are land extensions that are saturated or covered with fresh, marine, brackish, or brine water the entire year or part of the year. These land extensions occur in either the coastal zone or inland, being characterized by a high degree of biological interactions dependent on the water sources and high productivity.

There are several definitions for the term wetland. According to the Ramsar Convention on Wetlands of International Importance (www.ramsar.org), wetlands are "areas of marsh, fen, peatland, or water whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six meters." According to the US Environmental Protection Agency (water.epa.gov), the Clean Water Act defines the term wetland as "those areas that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs and similar areas." Wetlands have been classified into seven types (Table 1), three corresponding to coastal wetlands and four to inland wetlands (Mitsch and Gosselink, 2000).

Wetlands, Table 1 Wetland types according to Mitsch and Gosselink (2000)

Coastal wetlands	Inland wetlands
Tidal salt marshes	Freshwater marshes
Tidal freshwater marshes	Peatlands
Mangrove wetlands	Freshwater swamps
	Riparian systems

Introduction

Wetlands are ubiquitous, highly productive ecosystems occurring from tropical to polar latitudes on every continent except Antarctica. There are two main types of wetlands: (1) coastal or tidal wetlands and (2) inland or riparian wetlands. The latter are nontidal wetlands located on the edges of streams, lakes, or on low-lying lands where the accumulation of groundwater or precipitation saturates or covers the soil. These wetlands can be seasonal or perennial; nevertheless, in both cases their existence is of great importance for wildlife such as waterfowl that use them as feeding or breeding grounds (Mitsch and Gosselink, 2000). Common vegetation of these wetlands includes herbaceous plants, shrubs, and trees.

Tidal wetlands are at the boundary between freshwater and marine environments and closely coupled to estuaries. Similar to nontidal wetlands, these systems provide multiple ecosystem functions or services such as the capture of nutrients, sediment traps, carbon sinks, water filtering, food, shelter, fishery grounds, etc. Tidal action drives the mixing of fresh water and seawater, creating harsh environmental conditions for the development of vegetation; therefore, only a small group of plants have been able to develop physiological strategies to thrive in these habitats (halophytes, mangroves, sea grasses). However, there are coastal wetlands upstream where the influence of marine water is minimal, and the conditions are less severe for plant growth.

There is a wide variety of wetlands around the world, depending on the type of vegetation, amount of water present during the year, type of sediment, redox condition, amount of organic matter, and dominant vegetation. The different terms applied to wetland-like environments depend on the region of the world, mainly because the type of vegetation varies from one environment to another and the local names of plants and animals are not the same. For example, in North America the term "swamp" refers to a wetland with abundant wood plants (trees) present. This term cannot be applied in northeastern Europe, where there are no forested wetlands. Peat-accumulating wetlands are common in this part of the world, and the term "bog" is of common use. In addition, European wetlands have an abundance of nonwoody plants called "reed grass," giving rise to the name "reedswamps" for these systems. In Australia, the term "billabong" is used for wetlands formed from river meanders (Shiel, 1994). In North America, this same type of wetland is known as oxbow.

World diversity

European wetlands

In the Mediterranean Sea, the Rhone River Delta supports one of the most extensive wetlands in Europe, the Camargue, with an area ~9,000 ha. This wetland is a major nesting site for flamingos. Formerly, this site was used as rice fields (Mesléard et al., 1995). The Rhine River flows into the North Sea, and its delta is expansive. Currently, 7 % of this country (The Netherlands) is registered in the Ramsar Convention as a wetland of international importance (Wolff, 1993). In these wetlands, 250 waterfowl species can be found, including herons, cormorants, and spoonbills. Some of the most significant wetlands include the Oostvaardersplassen, which was artificially created 40 years ago as a wildlife sanctuary (Mitsch and Gosselink, 2000). The Wadden Sea has an extension of 8,000 km^2 of shallow waters, abundant mud flats, and marshes; it supports most of the fisheries from the North Sea, 60 % of brown shrimp, 80 % of plaice, and almost all herring (Maltby, 1986; Dugan, 1993). In the Baltic Sea, there are also important wetlands, and some of them are impacted by eutrophication due to nitrogen inputs from agricultural activities. This was the case in Langholm Bay (southwest Sweden), where macro- and microalgal blooms contributed to major losses of oxygen (hypoxia) during the 1970s and 1980s (Fleischer et al., 1994). Restoration efforts have been proposed to reduce the amount of nutrients entering from the inland (Kessler and Jansson, 1994). In Europe, there are important nontidal wetlands like those found along the Danube River and the Volga River. Unfortunately, both rivers have experienced environmental degradation as a consequence of drainage, agricultural activities, dam construction, and industrial pollution (IUCN, 1993).

African wetlands

Some of the wetlands in this continent are larger than those found in Europe, examples being the Inner Nigger Delta of Mali (320,000 km^2), the Zaire swamps (200,000 km^2), and the Sudd of the Upper Nile (~30,000 km^2) (Denny, 1993; Dugan, 1993). Fauna supported by these wetlands include elephants, crocodiles, buffalos, and many bird and fish species. Even indigenous tribes benefit from the shelter provided by these vast wetlands, as do urban populations, like the town of Maun (Mitsch and Gosselink, 2000). Some floras found in these wetlands include hyacinth (*Eichhornia crassipes*), kariba weed (*Salvinia molesta*), papyrus (*Cyperus papyrus*), and cattail (*Typha domingensis*). African wetlands can be classified as swamps according to European terminology and as marshes according to North American terminology. In the Ngorongoro crater, there are three exceptional wetlands: Mandusi Swamp, Gorigor Swamp, and Lake Makat. Wildlife is abundant in all of these marshes (Hanby and Bygott, 1998).

Australian wetlands

Seasons of dry and wet conditions produce wetlands that are intermittent along the east and west coasts of Australia (McComb and Lake, 1990). As noted above, the most representative Australian wetland is the billabong, an intermittent feature developed from an overflowing river channel (Shiel, 1994). One species of flora characteristic of the billabong is the river red gum (*Eucalyptus camaldulensis*), which surrounds these wetlands and provides habitat for many species of birds and fish. There has been severe loss of wetlands on this continent as well; along the west coast of Australia, 75 % of the wetlands in the Swan Coastal Plain has been lost (Chambers and McComb, 1994). Wetlands in New Zealand have experienced an even greater loss (90 %) of its former coverage. New Zealand possesses environmental conditions which are favorable for the development of wetlands, with an annual rainfall of up to 10 m (Mitsch and Gosselink, 2000). Common floral species in these wetlands are raupo (*Typha orientalis*), flax (*Phormium tenax*), and kahikatea (*Dacrycarpus dacrydioides*).

Asian wetlands

Most wetlands in this part of the world are located in Bangladesh, China, India, Indonesia, Myanmar, Papua New Guinea, and Vietnam. Water supply for these rivers and deltas originates in the highest parts of the Himalayas, flowing into a water delta of 600,000 km^2. In central Russia, the Ob River Estuary is one major site for waterfowl breeding (Dugan, 1993). Agricultural intensification, industrialization, deforestation, and damming are human activities that degrade the quality of these ecosystems (Beilfuss and Barzen, 1994). The Mekong Delta in Vietnam provides relevant ecosystem services as a source of fuel, medicine, and fisheries, providing more than 50 % of the protein requirements for the 20 million people living in the delta (Maltby, 1986). There are efforts to restore the Mekong Delta, as evidenced by a project known as Plain of Reeds (Beilfuss and Barzen, 1994). China has the largest extensions of Asian wetlands, covering more than 600,000 km^2. Most of them provide food, habitat, and recreation. Forty percent of these wetlands are included in the Ramsar list as systems of international importance (Parish and Elliot, 1990). The Yangtze, Zhujiang, and Liaohe rivers have the most important wetlands, but they are also the most populated areas. Some wetlands in these areas have been converted to rice paddies, reed fields, or fish ponds (Lu, 1990; Ma and Yan, 1989). Wetlands in China serve as refuge and migratory corridors; more than 160 species of birds have been documented in these systems (Lu, 1990).

American wetlands

In South America, river discharges into the Pacific are less than in the Caribbean; therefore, wetlands are seasonal along the Pacific. An example is the Palo Verde National Park in Costa Rica which has had diminishing diversity because of invasion by cattail (*Typha domingensis*) that has displaced native types of vegetation (McCoy and Rodriguez, 1994). The Orinoco River in Venezuela supports the Llanos wetland, the largest inland wetland in South America (Dugan, 1993). Flora in this wetland is the savanna type as a result of the dry summer seasons, unlike the flora found in the Orinoco Delta (Junk, 1993). Fauna characteristic of this zone are caiman (*Caiman* sp.) and red piranha (*Serrasalmus nattereri*). Pantanal in Paraguay is an inland wetland with an extension of more than 130,000 km^2 that is flooded each year (Hamilton et al., 1996). This wetland provides shelter and food to more than 600 species of birds, including ibis, egrets, the jabiru (*Jabiru mycteria*), emblem of Pantanal wetland (Por, 1995), and capybara (*Hydrochoerus hydrochaeris*) the largest rodent. The Amazon River has more than 300,000 km^2 of flooding forests. It is also subject to great pressure from land clearing activities, threatening the survival and habitat quality of vast wetland areas (Junk, 1993). Some of the major wetlands in North America include the Mississippi River Delta, Hudson Bay Lowlands, the Everglades, Prairie Potholes, Magdalena Bay, Marismas Nacionales, Ría Lagartos, and Sian Ka'an (Mitsch and Gosselink, 2000; CONABIO, 2009).

Types of wetlands

Tidal salt marshes are most abundant in temperate and polar latitudes. Flora characteristic of these wetlands include halophytes, plants that are tolerant of high salinities and partial submergence in marine water during the tidal cycle; the genera *Spartina*, *Juncus*, and *Salicornia* are examples. Tidal freshwater marshes are the transition between coastal and freshwater wetlands. They are distant from the coast but still influenced by the tides; however, they do lack the stress associated with high salinities found at the coast. Perennial and annual grasses and aquatic plants dominate this ecosystem (*Typha* sp., *Phragmites* sp., *Juncus* sp., *Hibiscus* sp.). In tropical and subtropical latitudes, tidal salt marshes are replaced by mangroves, a group of trees that have developed physiological and morphological strategies to thrive in saline to hypersaline environments (Hogarth, 2007). In America, mangroves are characterized by the genera *Rhizophora* (red mangrove), *Avicennia* (black mangrove), and *Laguncularia* (white mangrove). Mangroves in the tropics form tall canopies, while in the subtropics the trees are much smaller. The northern limit of mangroves along the Pacific coast of the USA is 29°N in Bahia de Los Angeles in the Gulf of California, while along the Atlantic Coast, the northern limit for mangroves is 27°N in Florida (Domínguez-Cadena et al., 2011). Smaller and less-dense mangrove forests are found at their northwest limit along the Pacific. Factors involved in this distribution are the lack of permanent rivers or streams, scarce rainfall, and low winter temperatures found in the Baja California Peninsula (Domínguez-Cadena et al., 2011).

Freshwater marshes are composed of grasses, sedges (Cyperaceae), pickerelweed (*Pontederia* sp.), and arrowheads (*Sagittaria* sp.). They have low amounts of peat, which is one of the key differences between freshwater marshes and peatlands. The latter have the largest deposits of peat, formed during the last glaciation. Bogs and fens are two types of peatlands occurring in lake basins; they lack large inflows or outflows, thus generating stagnant hypoxic conditions. In peatlands, mosses like *Sphagnum* sp. are common, as are carnivorous plants (*Cephalotus* sp.). Unlike freshwater swamps, the aforementioned inland wetlands (freshwater marshes and peatlands) lack woody vegetation. Freshwater swamps are forested wetlands with water present nearly the entire year. Cypress (*Taxodium* sp.) and tupelo (*Nyssa* sp.) are commonly found here (Mitsch and Gosselink, 2000). Riparian systems are found along rivers and streams where flooding occurs periodically during part of the year. They are rich in woody vegetation, including oak (*Quercus* sp.), cedar (*Chamaecyparis* sp.), and redwood (*Sequoia* sp.).

Summary

To understand the functioning of a wetland, it is necessary to consider several aspects: (1) the hydrology of the system, (2) the variation of the water level throughout the year, (3) the flow and its frequency whether perennial or intermittent, and (4) the influence of the tides, if the wetland is close enough to the coast. The climatic condition of a region is critical for the determination of the hydrologic characteristic of the wetland, as is the basin for its geomorphology. The geologic origin of the basin affects the steepness of the wetland. The amount of water that a wetland contains is less for systems located on steeper terrains which also are more likely to have coarser sediments. River deltas with nearly flat slopes are more likely to harbor fine-grained wetlands. Another crucial aspect in the hydrogeomorphology of a wetland is the physical environment, the chemistry of the soil (rich in iron-magnesium minerals or rich in carbonate), redox potential, amount of organic matter, salinity, pH, and amount of dissolved oxygen. Finally, there are biotic components of importance – bacteria, flora, and fauna – in the wetland that are influenced by the hydrological and the physicochemical characteristics. These organisms in turn modify the abiotic conditions of the environment by their activities (feeding, burrowing, excretion). The fluxes that occur between the biota and habitat make wetlands some of the most productive environments on Earth. The downside is that over 70 % of the human population of the world lives on or near coastlines, which places the greatest pressure on coastal wetlands. More information must be disseminated on the many services provided by wetlands to humans inhabiting the coastal zone.

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Cross-references

Anaerobic Environments
Anoxia, Hypoxia, and Dead Zones
Coastal Wetlands
Eutrophication
Mangroves
Wetlands Reclamation

WETLANDS RECLAMATION

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Synonyms

Wetland rehabilitation; Wetland renewal; Wetland restoration

Definition

Wetlands are seasonal or permanent freshwater- or saltwater-saturated dynamic ecosystems. Reclamation restores degraded wetlands to their former hydrological and ecological functions. Wetland reclamation is the rehabilitation of degraded wetlands habitat, reconstructing the impacted habitat as closely as possible to its original condition with respect to hydrological, ecological (vegetation, habitats), and morphological (soil) components.

Description

Wetlands are altered by the removal of stream bank vegetation, dredging and dredge-spoil disposal, ditching, intrusion of invasive species, and other factors. Wetland reclamation initially involves the stabilization of the degraded area, characterizing soil and water quality, and defining the inherent ecological features of the habitat.

Wetland reclamation activities require multidisciplinary expertise, including hydrologists, ecologists, geologists, engineers, etc. A successful wetland reclamation project must define how wetlands work, if the wetlands can be restored and created, and the best approaches to restore and create the wetlands (Mitsch, 2005). Elements common to wetland reclamation projects are site-selection criteria, hydrologic analysis, water source and quality, substrate augmentation and handling, plant material selection and handling, buffer zone placement, and long-term management (Kentula, 2002).

Estuarine wetlands are strongly influenced by tides; therefore, the dominant water source is tidal fresh or brackish water controlled by tidal action. In estuarine wetlands reclamation, restoration can be effectively accomplished by blocking channels which allow tidal saltwater inundation (USDA, 2008).

Similar to other reclamation projects in estuarine systems, wetlands reclamation in the early planning stage considers financial, social, and scientific feasibility studies that must be completed to determine whether the proposed reclamation activity is sustainable. The proposed goals must be monitored before, during, and after the reclamation project. Adaptive management is defined as the additional actions or adjustments that may be needed during the post-project monitoring process (USEPA, 2000).

Restoring lost or degraded wetlands is a valuable and cost-effective way for society to enhance wetlands ecosystem services. This includes reducing the risk to humans of impacts from coastal storms and other extreme events, improving food and water security, and increasing the capacity to mitigate and adapt to climate change (Alexander and McInnes, 2012).

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Cross-references

Coastal Wetlands
Marsh Drowning
Wetlands

WRACK

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Definition

The accumulation of dead plant matter in marine systems.

Description

In some productive marine ecosystems, plant matter accumulates at a faster rate than it can decompose. Dead plant material floats and aggregates at the landward margin of a marsh or beach or against any other barriers that hinder movement on the tide. At the high water mark, or strandline, wrack can form a mat centimeters thick that can block light and retain moisture. In temperate coastal wetlands, wrack mat formation has a seasonal cycle; the aboveground portion of plants senesces in the winter, leaving brittle, dry stems, which are easily broken by the elements and which form into wrack mats on the high tides in spring (Valiela and Rietsma, 1995). Seagrass, kelp, and macroalgae beds also produce wrack. Wrack mats are a disturbance that can kill underlying plants creating bare space in vegetated habitats (Bertness and Ellison, 1987). Where stressful edaphic conditions are exacerbated by the loss of the plant canopy, as in salt marshes, bare spots formed by wrack mats can be persistent and are colonized by stress-tolerant fugitive species (Bertness, 1991). However, some animals seek refuge in the cooler and moister microclimate of wrack mats, and wrack is an important food resource for detritivores, particularly in low productivity systems like beaches (Dugan et al., 2003).

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Cross-references

Detritus Food Webs
Macroalgae
Macroalgal Blooms

Z

ZOOPLANKTON

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Definition

Zooplankton are small animals suspended in the water that are neither attached to the bottom nor able to swim effectively against most currents. Holoplankton taxa spend all life stages in the plankton. Examples of permanent members of the zooplankton include copepods, rotifers, cladocerans, ctenophores, and chaetognaths. Meroplankton include taxa in which only certain life stages are planktonic (e.g., larvae of crustaceans, annelids, molluscs, tunicates, and fishes or adults such as medusa).

Characteristics

Zooplankton are ubiquitous and omnipresent in estuaries worldwide. A short, small mesh zooplankton collection in an estuary can yield many thousands of individuals representing dozens to hundreds of species and life stages, many of which have not yet been described (Johnson and Allen, 2012). Abundances tend to be highest in temperate estuaries, in intermediate to high salinity zones, and during the warmest months of the year. Higher diversity and abundance occur near structured habitats such as sea grass beds, mangroves, salt marshes, and oyster reefs. Many commercially important molluscs, crustaceans, and fishes depend on the completion of planktonic stages of development within estuaries.

The size range of zooplankton spans more than four orders of magnitude. The smallest size group includes heterotrophic flagellates and rotifers. Midsized zooplankton (0.2-2 mm) include copepods and larvae of familiar groups of invertebrates (e.g., barnacles, polychaetes, bivalves, and echinoderms). Larger zooplankton (2-20 mm) include shrimp and crab larvae, peracarid crustaceans (e.g., mysids, amphipods), chaetognaths, and larval fishes. The largest members, large jellyfishes (scyphomedusae) and siphonophores, can exceed 1 m.

Motility in zooplankton is usually underestimated, but, when considered relative to their size, swimming speeds often exceed those of vertebrates. Small ciliated forms move slowly. Crustaceans (including copepods) with jointed appendages, chaetognaths and medusae with simple muscular contractions and larval fishes with fins can attack prey or change positions in the water column. In estuaries, some zooplankton use behavioral responses to light, salinity, and tidal currents to move and maximize retention in or export from estuaries. Sophisticated escape responses allow many species to avoid predators or collection with towed nets. Additional antipredator defenses include transparency, protective spines, bioluminescence, distasteful or toxic chemicals, diel activity patterns, and synchronized mass spawning.

Zooplankton play a central role in estuarine trophic dynamics, providing a critical link between phytoplankton and bacterial productivity as they transport energy to higher trophic levels including some large fishes. Some also consume detritus (often from land-based sources) and help incorporate this energy source into the food web. Estuarine zooplankton-based energy can also be exported to the ocean.

M.J. Kennish (ed.), *Encyclopedia of Estuaries*, DOI 10.1007/978-94-017-8801-4,

Zooplankton are considered very good indicators of environmental change because they have short generation times and limited mobility. Reports of shifts in distributions and phenology have been attributed to climate change. In recent years, zooplankton assemblages in many estuaries worldwide include exotic species introduced from across the globe often as a result of transport by commercial shipping.

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