DEVELOPMENTS IN
ECOSYSTEMS

1

WETLANDS ECOSYSTEMS IN ASIA: FUNCTION AND MANAGEMENT

EDITED BY

M.H. WONG

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CROUCHER INSTITUTE FOR ENVIRONMENTAL SCIENCES

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Abridged Contents

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Contents

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Preface

Compared to other ecosystems, wetlands have received an exceptional amount of attention. Wetlands are valuable as sources, sink and transformers of a multitude of chemical, biological and genetic materials. They stabilize water supplies, clean polluted waters, protect shorelines, and recharge groundwater aquifers. They have increasingly become recognized for their unique ecological functions in the environment and are the focus of increased research by scientists and study programs by schools, communities, and nature centers. On the other hand, the idea of using constructed wetlands for wastewater treatment has been encouraging because of their environmental friendliness and enhancement on landscape quality. Consequently, interest in wetlands extends from students in landscape architecture and environmental engineering programs to the real world of public officials, developers, and private citizens.

Wetland management requires an understanding of the scientific aspects of wetland balanced with legal institutional and economic realities. This book consists of comprehensive information of wetland's importance, functions, conservation and management strategies, which will be beneficial to environmental professionals in different fields for formulating wetland conservation policy and conducting environmental research. The latest and advanced information and management techniques of using constructed wetland for wastewater treatment are also included in this book.

This book is the product of the Croucher Advanced Study Institute on Wetland Ecosystems in Asia: Function and Management held in March 2003 at Hong Kong Baptist University, attended by a selected number of specialists and practitioners, to review the major problems involved in wetland management, and how they can be solved, against a background of situations in Asian countries. The Asian region contains some of the world's largest and diverse seagrass beds and about half of the approximately 50 seagrass species known world-wide occur along Asian coasts. These seagrass beds, mainly via the detritus food chain, support a very productive community of fish and invertebrates, especially mollusks and crustaceans, many of which are of commercial importance. The South-East Asian peat swamp forests cover nearly 30 million hectares compared to only one million hectares in Amazonia. Asia's major rivers (a wetland habitat) are some of the world's largest and most of the rivers of Asia have extensive floodplain wetlands. The region also contains the world's largest contiguous area of mangroves – the Sundarbans in Bangladesh, and the country with the world's largest expanse of mangroves – Indonesia. It is also global center for mangrove diversity and evolution. In terms of freshwater ecosystems, the swamp forests of South-East Asia are not only among the largest and the best developed in the world, but are botanically among the most diverse, while exhibiting a high degree of endemism.

Unfortunately, wetlands throughout Asia are under threat, destruction and degradation continues unabated. Analysis showed that of nearly 1,000 wetlands considered to be of international importance for socio-economic or biodiversity values in Asia, as many as 56% were considered to be moderately or seriously threatened, while only 15% were threatened. In addition, only about 10% of these internationally important wetlands are currently totally protected, while a further 15% is partially protected.

To date, in South-East Asia, 5 countries have developed their own National Wetland Policy or Wetland Action Plan or National Wetland Strategy. They are Indonesia, Philippines, Vietnam, Thailand and Cambodia. This book discussing different wetland management strategies in Asia will act as a reference book for environmental professionals in other Asian countries to formulate conservation policy for their own countries.

The book consists of 4 sessions: I. Natural Wetland Systems and Their Functions; II. Wetland Biogeochemisty; III. Wetland Management Strategies in Asia and IV. Constructed Wetlands. The basic information of natural wetland systems is introduced in Session I. More scientific discussion about the biogeochemistry of wetland can be found in Session II. In Session III, wetland management strategies of different Asian countries, including Malaysia, the Philippines, Vietnam, Thailand and Hong Kong are discussed. Although only Asian experience has been shared, past experience shows that there is a body of general rules applicable to different wetland systems of different countries. The latest and advanced information and management techniques of constructed wetlands can be found in Session IV which will be useful for environmental managers and engineers working on constructed wetland projects.

We hope that this book will not only be beneficial to environmental professionals for formulating wetland conservation policy and conducting environmental research, but it will also serve as a reference book for students of undergraduate and graduate courses on ecology and conservation.

About the Editor

Professor Ming H. Wong

After graduation from the Chinese University of Hong Kong with a BSc in Biology, Professor Wong obtained his MSc, PhD and DSc from the University of Durham and also MBA and DSc from the University of Strathclyde. Professor Wong served the Biology Department of The Chinese University of Hong Kong as Lecturer and Senior Lecturer from 1973–85, following which he became Head of the Biology Department of Hong Kong Baptist University (1986–2002) and was promoted to Chair Professor in 1990. He currently serves as Director of the Croucher Institute for Environmental Sciences, Hong Kong.

Professor Wong's research work centers around restoration of derelict land and pollution ecology, especially heavy metals in earlier years, and persistent toxic substances more recently. He serves as regional co-ordinator of Central and North East Asia for the project "Regionally based assessment of persistent toxic substances" sponsored by the Global Environmental Facility (GEF) and implemented by the United Nations Environment Programme (UNEP).

Professor Wong has over 200 papers published in international scientific journals and edited several books. He serves on editorial boards of eight scientific journals related to environmental science, including Editor-in-Chief of the journal "Environmental Geochemistry and Health (Kluwer Academic Press)" and is a Visiting Professor of several major institutes in Mainland China such as Nanjing Institute of Soil Science of The Chinese Academy of Science, Zheijian University, Wuhan University, and Zhongshan University, and also Middlesex University in the UK.

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Session I

Natural Wetland Systems and Their Functions

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Developments in Ecosystems, volume 1
Edited by M.H. Wong

Chapter 1

A Comparison of Issues and Management Approaches in Moreton Bay, Australia and Chesapeake Bay, USA

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Abstract. Management of coastal systems is becoming increasingly important, however understanding the process of effective management often remains elusive. This chapter contrasts examples of environmental problems and associated management in Moreton Bay, Australia, and Chesapeake Bay, USA. Targeted research in Moreton Bay identified specific issues which led to changed practices, while intense management and research in Chesapeake Bay has been unable to keep pace with increasing anthropogenic stress. The balance of political, financial and scientific aspects of a management solution is discussed, with global examples. Sustainable solutions to environmental problems in coastal ecosystems will only be achieved with a rigorous approach to management and the development of global standards.

1.1. Introduction

As humans continue to impact coastal ecosystems at a global scale, coastal management can be viewed as a globally significant and important activity (IGBP, 2001). Coastal management can be considered to be the sum total of human interactions within an ecosystem, whether or not these interactions are formalized into a management structure or series of documents. Accepting this assumption, coastal management is a major environmental issue for the globe, involving more people in more ways than many other issues. Management of the coastal zone is typically complicated, involving multiple jurisdictional boundaries and a variety of issues. In most cases, the plethora of human activities are not encompassed into a coastal management structure, rather they evolve around various issues and activities that impinge on coastal management. Thus, coastal management

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activities are often not well documented and developing global data sets regarding coastal management issues is difficult. This chapter describes two main case studies in order to draw out the issues of environmental problem solving. These case studies serve to illustrate the point that each environmental problem can benefit from scientific research, and a solution-focused management approach can be developed for each problem in collaboration with the community. The problems, research, and solution-focused management approaches presented for each case study are in no way comprehensive — there are many more problems, more research and more solutions than covered here.

1.2. Comparison of Systems

The two principal case studies are Moreton Bay, Australia, and Chesapeake Bay, USA. In many respects, Moreton Bay is approximately one tenth of Chesapeake Bay (Fig. 1, Table 1). In terms of human population, Moreton Bay has roughly

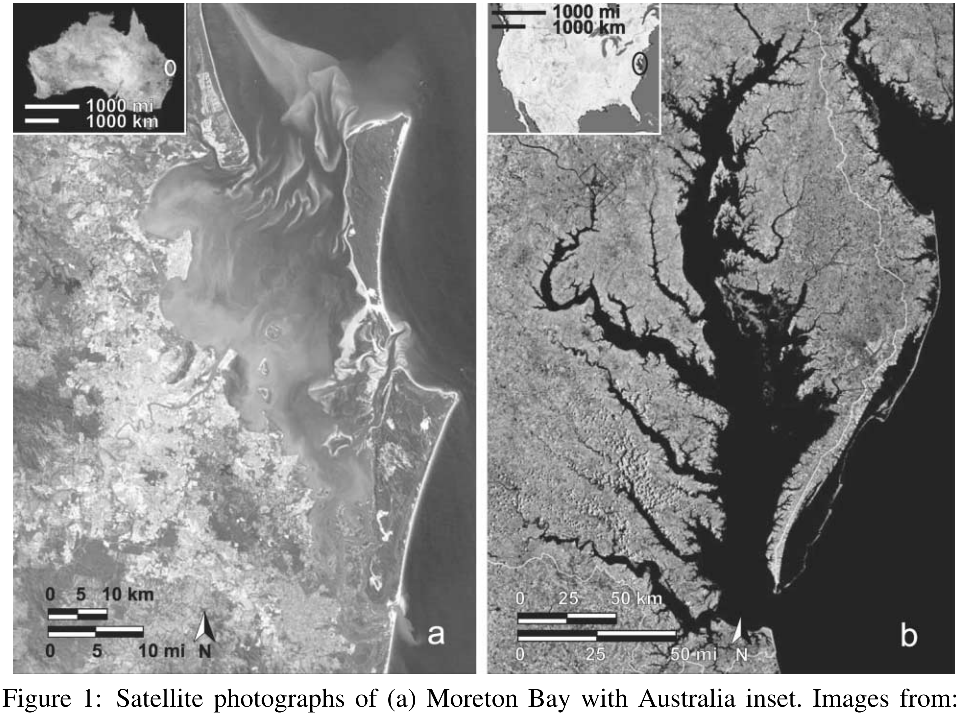

Figure 1: Satellite photographs of (a) Moreton Bay with Australia inset. Images from: Australia, ACRES Landsat 7 Mosaic of Australia, Pseudo Natural Color Image; Moreton Bay, ACRES Landsat 7, 21 March 2003, Natural Color Image and (b) Chesapeake Bay with USA inset. Images from: USA, NASA visible earth; Chesapeake Bay, USGS.

Table 1: Comparison statistics for Moreton Bay and Chesapeake Bay.

	Moreton Bay	**Chesapeake Bay**
Latitude	27° S	38° N
Watershed area	21,220 km^2/8193 mile^2	165,800 km^2/64,000 mile^2
Bay area	1,523 km^2/588 mile^2	18,130 km^2/7000 mile^2
Watershed population	Approx. 1.5 million	Approx. 15 million
Average depth	6.8 m/22 ft	6.4 m/21 ft

Data from Dennison & Abal (1999), Horton (2003) and Skinner et al. (1998).

1.5 million people living in its watershed, mostly in the city of Brisbane, while Chesapeake Bay has roughly 15 million people, including the cities of Washington DC, Baltimore, Norfolk and Richmond. In terms of watershed area, the Moreton Bay watershed is ~21,000 km^2 while Chesapeake Bay is ~165,000 km^2. In terms of bay area, Moreton Bay is ~1,500 km^2 and Chesapeake Bay is ~18,000 km^2 (Skinner et al., 1998; Horton, 2003) (Table 1). Therefore, the ratio of people to bay are roughly proportional in both systems and so, in terms of population pressure and potential anthropogenic effects, Moreton Bay can be viewed as a microcosm of Chesapeake Bay.

Both bays are adjacent to industrialized, urban/suburban developments with a well developed management infrastructure. Both are situated on the east coast of a continent with a warm offshore current, have a mean depth of ~6–7 m, historically productive fisheries, a fringe of mangrove forest or salt marsh and historically extensive seagrass and oyster reefs. One important difference is that Moreton Bay is subtropical, located at 27° S, while Chesapeake Bay is temperate, located at 37° N.

The balance of environmental concerns differ between Moreton Bay and Chesapeake Bay, with pulsed sediments being the largest issue in Moreton Bay (and nutrients secondarily) while nutrients are the largest issue in Chesapeake Bay (and sediments secondarily) (Fig. 2, Table 2). Moreton Bay has one large connection to the sea, with two smaller entrances, while Chesapeake Bay only has one large sea opening (Fig. 1).

Both systems have been relatively well studied on a global scale. While Moreton Bay has had recent intensive research, Chesapeake Bay has had intensive research historically and recently, making it one of the most studied estuaries in the world (Tibbets et al., 1998; Dennison & Abal, 1999; Ernst, 2003). In both regions, a heightened awareness of bay issues has been developed with the aim of achieving protection and restoration. The development of protection is appropriate for Moreton Bay, while Chesapeake Bay requires a major restoration effort.

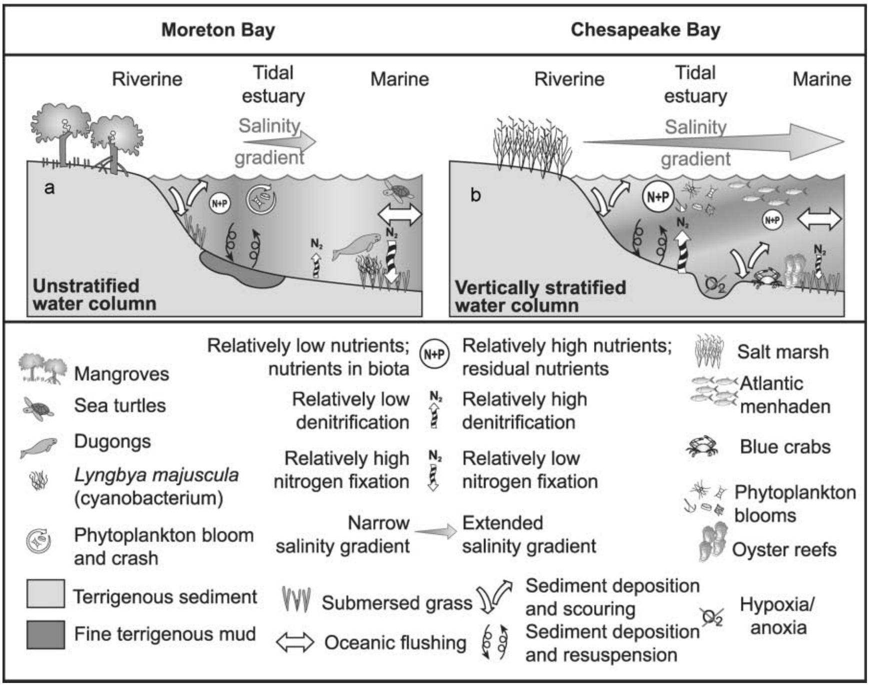

Figure 2: Conceptual diagrams contrasting the major features of (a) Moreton Bay and (b) Chesapeake Bay.

1.3. Moreton Bay Overview

Moreton Bay is fringed with mangroves and has two major rivers discharging into the western bay. The Brisbane and Logan Rivers have watersheds that extend to the Great Dividing Range, west of Brisbane. Rainfall in the Moreton Bay region is intermittent, with short intense rainfall interspersed with long periods of dry conditions. The highest rainfall events are associated with monsoonal depressions during summer (December–February) and sediment inputs occur during these pulsed river flow periods. Thus, the rivers only flow for a short time, and the tidal sections of these river-estuaries act as seawater inlets or coastal embayments for much of the year (Davies & Eyre, 1998; Carruthers et al., 2002). This results in a narrow salinity gradient during the predominant dry periods, extending only tens of kilometers within the river/ estuary. The bay itself retains full strength salinity for most of the year, but during high rainfall periods significant reductions in bay salinity can occur. Moreton Bay experiences a 1.7 m tidal range, and the ensuing mixing combined

Table 2: Examples of environmental problems from Moreton Bay and Chesapeake Bay.

Estuary	Problem	Result	Research results	Potential solutions
Moreton Bay	Fine grained sediments	Seagrass loss	Sediment from channel erosion in agricultural regions	Replant and fence eroding channels
	Sewage nutrients	Macroalgal blooms	Sewage plumes mapped	Biological nutrient removal upgrades
	Lyngbya blooms	Human health issues	*Lyngbya* blooms linked to forestry practice	Monitoring and revised forestry practice
Chesapeake Bay	Nutrient addition	Hypoxia/ anoxia	Decomposing phytoplankton lead to oxygen depletion	Reduction of point and diffuse nutrient sources
	Critical habitat loss	Oyster and seagrass loss	There are multiple causes of decline	Oyster restocking and seagrass restoration
	Accelerated erosion	Sedimentation	Shoreline erosion influenced by sea level rise	Augment marshes and islands, e.g. possible use of dredge spoil
	Harmful algal blooms	Fish kills, human health, hypoxia	Nutrients and salinity important, also nutrient interactions	Nutrient reductions, continuous nutrient monitoring

List of problem statement, research findings and management solutions either proposed or enacted.

with the lack of freshwater results in vertically unstratified water masses. In addition, wind driven mixing leads to sediment resuspension in the western margins of the Bay and the water is often brown in color (Longstaff et al., submitted). Low dissolved oxygen conditions are not common in Moreton Bay. Water circulation in the bay is generally in a clockwise direction, with onshore prevailing winds leading to the poorest flushing in the western embayments near the river mouths (Fig. 3a). Nutrients derived from both point and non-point sources are delivered primarily into the western bays and strong horizontal gradients exist for most water quality parameters. The nutrient inputs into Moreton Bay are rapidly assimilated by biota or deposited into sediments, such that water column nutrients in the bay are near detection limits most of the time (O'Donohue et al., 2000). Moreton Bay has an assemblage of tropical seagrass that support a large population of dugong and sea turtles. Recent outbreaks of a harmful algal bloom (*Lyngbya majuscula*) have occurred in this area where there is a large trawl fishery for penaeid shrimp and intensive recreational fishing.

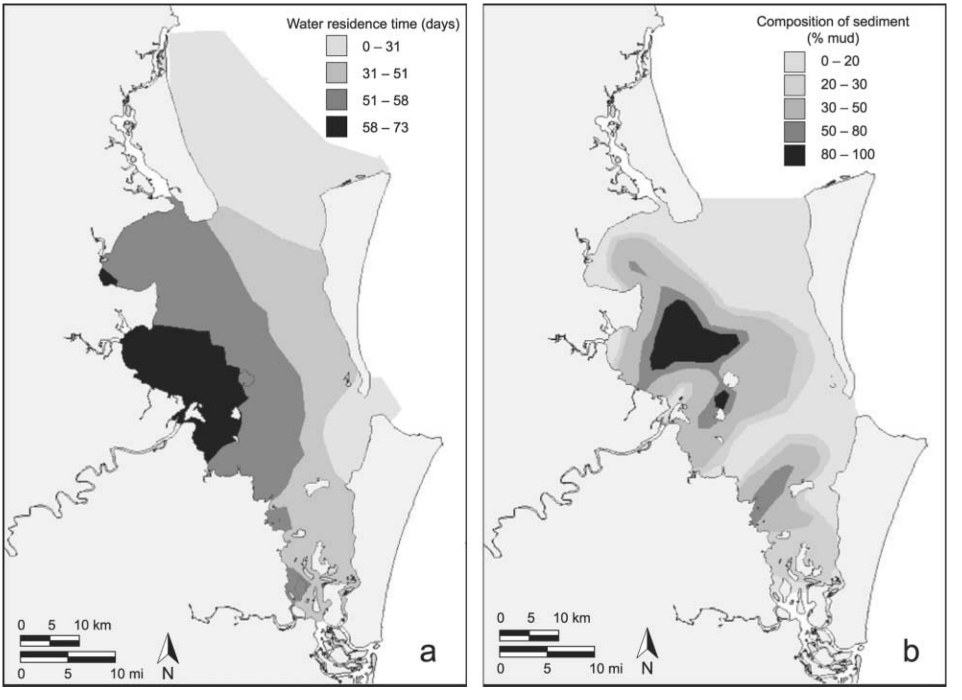

Figure 3: (a) Residence time of water in Moreton Bay Australia, data from Longstaff et al. (2004). (b) Sediment type throughout Moreton Bay, Australia, data from Longstaff et al. (2004).

1.4. Moreton Bay Sediments and Seagrass Loss

Problem The watershed of Moreton Bay is sparsely vegetated (due to clearing for agriculture and urban development) and large rainfall events deliver sediments into the rivers and eventually into the Bay (Table 2). These sediments are largely deposited into the deep (10–20 m) basin in western and central Moreton Bay. The fine grained sediments form mud deposits that are frequently resuspended by the dominant southeast wind in the region (Longstaff et al., submitted) (Fig. 3b). The resuspended sediments increase water turbidity and reduce light penetration. As a result of reduced light penetration, seagrass growth is inhibited (Fig. 4a). Seagrass losses have been observed in the turbid regions of the bay, leading to loss of habitat for juvenile penaeid shrimp as well as loss of grazing areas for turtles and dugong (Abal & Dennison, 1996).

Research Once the problem of seagrass loss was recognized and linked to resuspension of fine-grained sediments, the scientific challenge was to locate

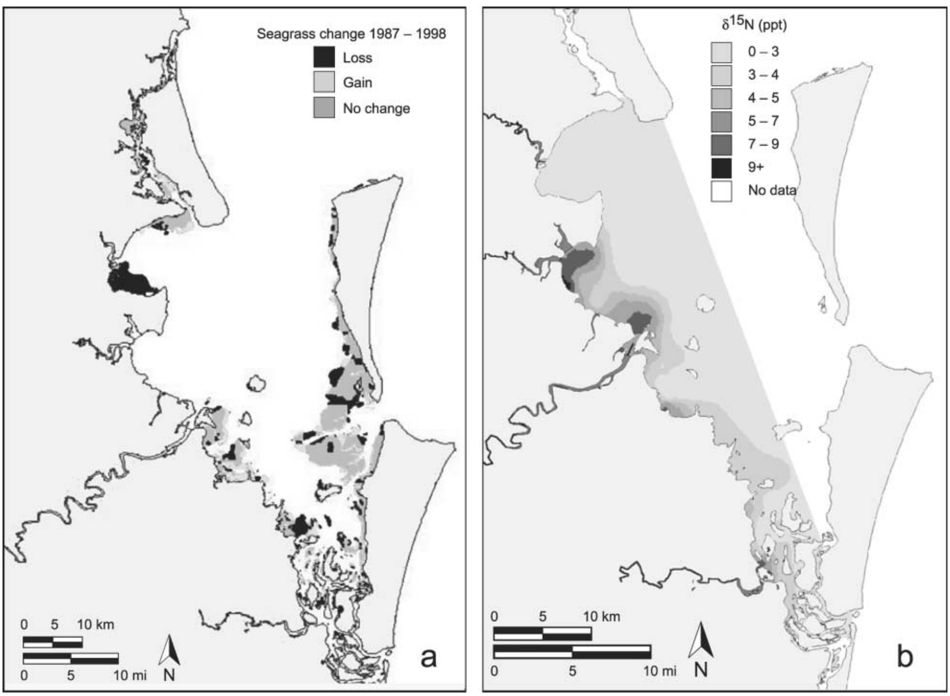

Figure 4: (a) Change in seagrass cover in Moreton Bay between 1987 and 1998, data from Longstaff et al. (submitted). (b) Sewage plume map ($\delta^{15}N$) for Moreton Bay in September 1997. After Costanzo et al. (2001).

the source(s) of these sediments in order to develop control measures. Levels of two geochemical tracers (thorium and lanthium) in Moreton Bay sediments were compared with various soil samples from the watershed. Sediment dating of cores taken from the central mud patch revealed that the fine grained sediments available for resuspension were deposited relatively recently (within the past 90 years). Thus, it was clear that human alterations of the watershed had accelerated natural processes of sedimentation. The sediment source was shown to be confined to subwatersheds of the Brisbane and Logan Rivers. In another set of tracer measurements, the amounts of radium and cesium in sediments of the Brisbane and Logan Rivers were compared with watershed topsoils (cultivated and uncultivated) and subsoil. These results indicated that land disturbance and, in particular, channel erosion was the principle mechanism of soil erosion contributing to Moreton Bay sediments. This channel or gully erosion in the smaller streams was being exacerbated by agricultural fields without riparian buffers next to streams and grazing activities of cattle and sheep which removed riparian vegetation and weakened stream banks.

Working toward a solution Once research had identified the highly erosion susceptible areas in the watershed, as well as the mechanism of erosion, it was possible to initiate a targeted approach for management actions (Table 2). The practical solution was to fence the livestock and prevent grazing activity while revegetating already degraded stream banks. Another important component of the solution was the education of land owners with regard to the linkage between land use and sediment runoff; this was done by community involvement in field trials of riparian revegetation.

1.5. Moreton Bay Sewage Plumes

Problem Nutrients entering Moreton Bay led to large beach wracks of macroalgae ("sea lettuce"-*Ulva* sp.) near the Brisbane River mouth and occasional dinoflagellate blooms in the western embayments (Uwins et al., 1998; Dennison & Abal, 1999) (Table 2). The majority of sewage effluent discharge occurs into the rivers, with 18 major ($>$0.5 ML of effluent per day) treatment plants discharging into the Brisbane and Pine Rivers alone (Dennison & Abal, 1999). Since these rivers are highly turbid and little biological processing of nutrients occurs, the river mouths discharge the bulk of the nutrients into the western bays of Moreton Bay (O'Donohue et al., 2000) (Fig. 4b). The extent and relative proportion of the sewage contribution to this nutrient over-enrichment problem was previously unknown.

Research A technique for tracing sewage plumes was developed using marine plants as biological indicators of nutrient sources. Marine plants readily absorb nutrients for growth and nutrition and the ratio of various naturally occurring isotopes of nitrogen in the plant tissue reflects the ratio in the surrounding water (Wada, 1980; Grice et al., 1996; Udy & Dennison, 1997; Dennison & Abal, 1999; Waldron et al., 2001). The ratio of ^{14}N:^{15}N, relative to an atmospheric standard (calculated as $\delta^{15}N$), is variable and different nitrogen sources have different $\delta^{15}N$ values. Preliminary investigations demonstrated that the $\delta^{15}N$ of a species of red macroalgae (*Catenella nipae*) would reflect the $\delta^{15}N$ signature from sewage nitrogen inputs within several days. The method involves deploying and retrieving several hundred macroalgae in a grid throughout the bay, with the resulting $\delta^{15}N$ values being spatially analyzed and mapped (Costanzo et al., 2000). These maps revealed distinct sewage plumes emanating from high input areas (Costanzo et al., 2001).

Working toward a solution The preparation and dissemination of sewage plume maps using the biological indicator results was an extremely powerful tool for stimulating sewage treatment upgrades in the region (Table 2). These upgrades, staged over several years and costing hundreds of millions of dollars, resulted in dramatic reductions in sewage plume extent and also reduced wracks of *Ulva* sp. in the vicinity of river mouths. Further improvements in sewage treatment technologies and increased wastewater reuse should continue the trend of sewage plume reductions. Reduction of known point sources of nutrients makes non-point nutrient inputs easier to identify and quantify. Reduction of these diffuse sources is the next challenge to solve.

1.6. Moreton Bay Harmful Algal Blooms

Problem Outbreaks of a marine cyanobacterium that caused human and ecosystem health problems began in the 1990s in Moreton Bay (Dennison et al., 1999) (Table 2). Moreton Bay fishermen began complaining of skin lesions as well as throat and eye irritation when an unusual proliferation of filamentous ‘weed’ covered the seagrass. Investigation revealed the presence of *Lyngbya majuscula*, a cyanobacterium with toxins known to cause contact dermatitis (Osborne et al., 2001). During the mid- and late-1990s, *Lyngbya* spread to other regions of the bay, smothering seagrass and mangroves, with large wracks washing up on swimming beaches (Fig. 5). Turtle and dugong populations appeared to be affected, tourism and fish catches have reduced and nitrogen inputs through *Lyngbya* nitrogen fixation may even counteract some of the nitrogen reduction strategies.

Research An intensive research program was initiated to determine the cause(s) of *Lyngbya* initiation and proliferation. Initial results pointed to the availability of dissolved iron as a trigger for this cyanobacterial bloom initiation. Subsequent research into the iron chemistry and runoff from various

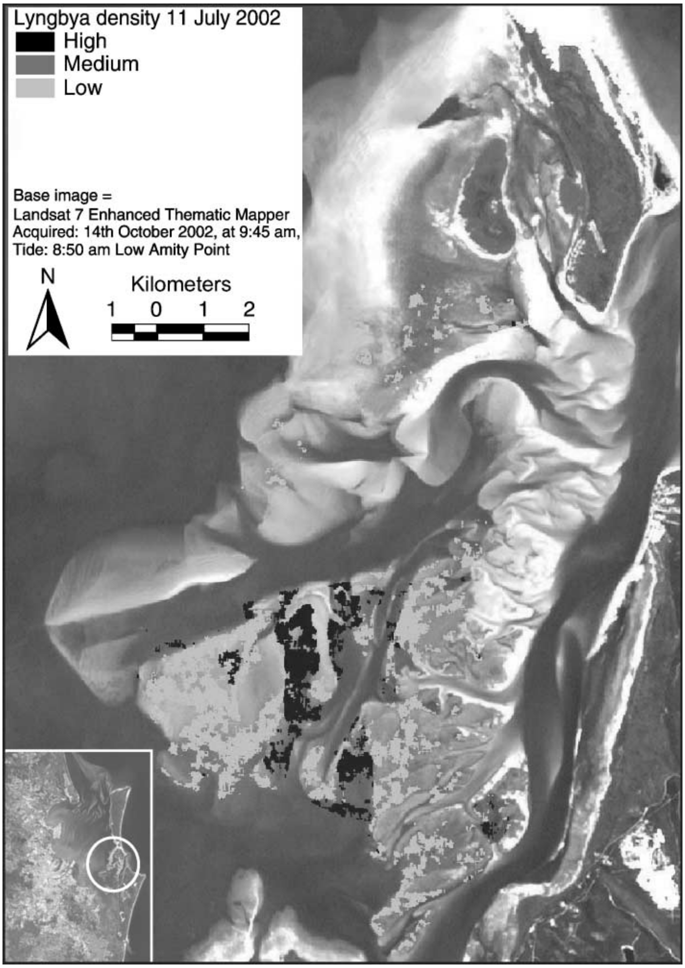

Figure 5: Distribution of *Lynbya majuscula* bloom in Moreton Bay during 2002, Data provided by C. Roelfsema, University of Queensland, Australia 2003.

potential land sources revealed a link between land clearing of plantation pines and runoff of organic-rich water containing dissolved iron and *Lyngbya* blooms. A phase of rapid deforestation in the 1990s due to rotation cycles, economic factors as well as a wildfire event in the pine plantations were hypothesized to result in pulses of organic-stained water into the bloom initiation region. The initial results of dissolved iron and *Lyngbya* stimulation, as well as observations of orange-stained water with high iron levels in the vicinity of canal estates, led to early suspicions that dredging and filling could be stimulating blooms. Eventually, the organic compounds in runoff from pine plantations were found to make the dissolved iron more bioavailable to the cyanobacteria and so this acidic, black water runoff enhanced *Lyngbya* growth rates (Albert, 2001; Rose & Waite, 2003).

Working toward a solution Various mechanical harvesting techniques were trialed, and were largely unsuccessful due to logistic and economic considerations. A moratorium on canal estate construction was discussed, but eventually discounted as runoff from pine plantation deforestation was thought to be the major problem (Table 2). A program involving the forestry industry, in which various trials of forestry practices will test rates of organic and iron-rich runoff was established. A predictive model is currently being developed that will be used to guide future management decisions.

1.7. Chesapeake Bay Overview

Chesapeake Bay is fringed with salt marshes and has several major rivers discharging into the western bay (Potomac, Rappahanok, James, York) that have their origins in the Appalachian Mountains in western Maryland, Virginia and West Virginia. However, the bulk of freshwater inputs are from the Susquehanna River which drains a large section of Pennsylvania and discharges into the northern bay. The salinity gradient is extensive (360 km along the main axis of the bay). Runoff is more or less continuous, forming a distinct salt wedge (Boicourt et al., 1999). Chesapeake Bay was formed as a drowned river valley of the Susquehanna River and low oxygen conditions occur in bottom waters, particularly in the deep trough formed from this historical valley. Chesapeake Bay has a minimal astronomical tidal range (<0.5 m), but meteorological tides due to weather patterns contribute to mixing. Water circulation in the bay is largely driven by the freshwater inflows with seawater extending further up the eastern shore due to the Coriolis force. Nutrients derived from both point and non-point sources are delivered throughout the bay, with predictions that agriculture inputs constitute 55.2% and sewage 20.7% of nitrogen export from the watershed into Chesapeake Bay (Boynton et al., 1995; Castro & Driscoll, 2002). As a result, up to 27% of Chesapeake Bay has been classified as eutrophic (Kiddon et al., 2003). Phytoplankton blooms are common, including some toxic species. Residual nutrients remain in the water column throughout the bay for most of the year. Chesapeake Bay has an assemblage of freshwater and marine submersed grasses that support large waterfowl populations. There is a large crab, anchovy and menhaden fishery and intensive recreational fishing. Historically, there was a fishery based on the abundant anadromous shad and a massive oyster fishery.

1.8. Chesapeake Bay Nutrient Over-Enrichment

Problem A major problem in Chesapeake Bay is the development of summertime low dissolved oxygen in bottom waters, particularly in the deep basins (Table 2). While increased land clearing was related to increased hypoxia between 1700 and 1900, the advent of fertilizer use during the 20th century has led to unprecedented anoxic events since the 1970s (Cronin & Vann, 2003) (Fig. 6). This hypoxia (low oxygen) or anoxia (no oxygen) is detrimental to benthic organisms, including a reduction in oyster (*Crassostrea virginica*) growth rates (Widdows et al., 1989). In addition, anoxic bottom waters facilitate the release of sediment nutrients, particularly phosphorus and ammonium, mobilizing nutrients that would otherwise be locked up in sediments. The morphology of the bay and the naturally stratified water column results in a natural tendency for low oxygen in bottom waters. What has developed into a problem is the increase in severity, extent and persistence of the low oxygen events. The volume of bay water affected by hypoxia (dissolved oxygen $<2\ \mathrm{mg\ l^{-1}}$) has been steadily increasing since the 1950s (Cronin & Vann, 2003). In wet years (e.g. 1998), the

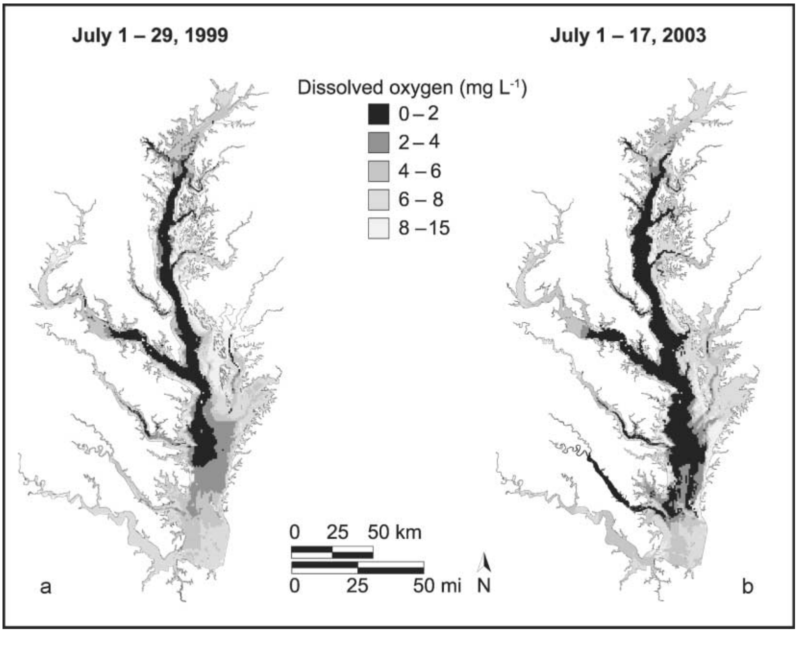

Figure 6: Dissolved Oxygen in Chesapeake Bay in (a) July 1999 (a standard year) and (b) July 2003 (a high rainfall year). Data provided by Chesapeake Bay Program.

hypoxic and anoxic waters threaten not only the deep basins, but also the more shallow regions.

Research Low oxygen bottom waters have been linked to the growth and subsequent decay of phytoplankton (Kemp et al., 1992). Phytoplankton growth is stimulated by nutrient inputs, particularly the winter/spring runoff that leads to a spring bloom of diatoms (Anderson et al., 2002). The microbial decay of diatoms as they settle to the bottom consumes oxygen faster than it can be replenished by diffusion and advection from the surface, leading to hypoxia and anoxia. A variety of measurements and models have been developed with relatively good predictive capacity. Even a basic model using only two variables to predict the hypoxic volume of the bay can be effective. The product of total nitrogen concentration and river flow at the tidal limit has been used successfully to predict hypoxic volume (Hagy, 2002).

Working toward a solution The reduction of nutrient point sources mandated throughout a multi-jurisdictional agreement (Chesapeake Bay Agreements of 1987 and 2000), has resulted in nutrient reductions in some locations. For example, the Patuxent River nutrient concentrations dating back to 1960 have been monitored, with significant reductions occurring as a result of sewage treatment upgrades (Table 2). These nutrient reductions are largely associated with western shore urban regions, while agricultural regions of the eastern shore showed significant increase in nutrient export between 1985 and 1995 (Glibert & Magnien, 2004). What the Patuxent River data demonstrates is that local government and community partnerships can accomplish real reductions in nutrient inputs. However, the broader problems of various diffuse sources including atmospheric inputs, agricultural runoff and septic inputs to Chesapeake Bay are showing no signs of abating.

1.9. Chesapeake Bay Critical Habitat Loss

Problems Historically, Chesapeake Bay supported extensive oyster reefs (Table 2). These reefs were built up from layers of dead oyster shells, with a top layer of live oysters. The reefs provided habitat for various attached organisms as well as a place for fish to congregate. The bay also supported vast meadows of submersed aquatic plants-extending from marine salinities to the fresh water river reaches. These aquatic grass beds provided habitat for juvenile fishes and invertebrates as well as food for some waterfowl (e.g. canvasback ducks). Over the last 30–40 years oyster populations have reduced to only 1% of historical abundance in the Chesapeake, while aquatic grasses occupy less than 10% or their historic distribution (Fig. 7a) (Rothschild et al., 1994; Boynton et al., 1996; Orth et al., 2002).

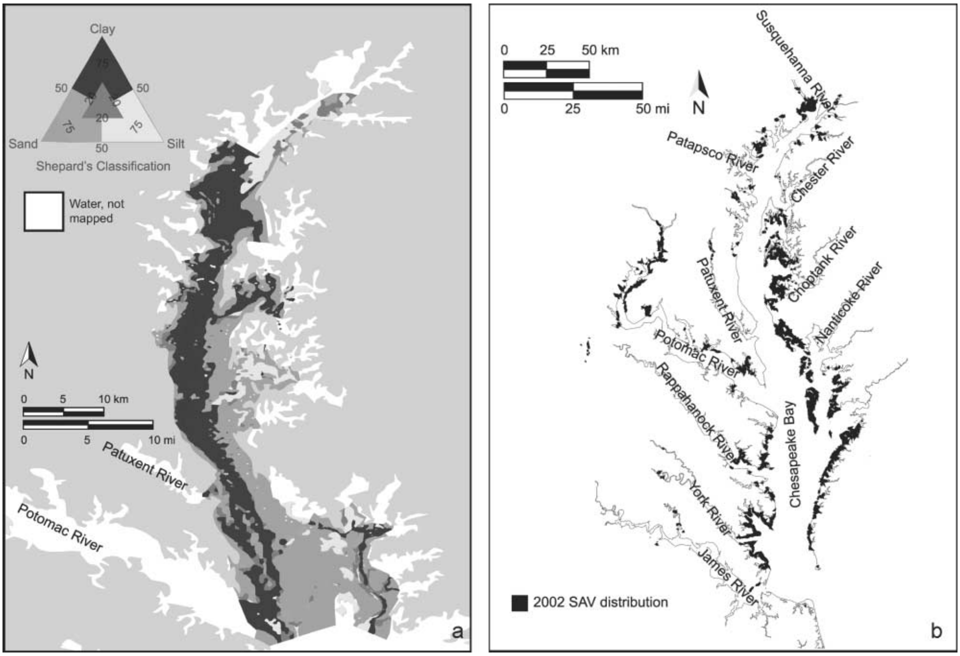

Figure 7: (a) Submersed grass (SAV) distribution in Chesapeake Bay. Data provided by Dave Wilcox, VIMS SAV lab and (b) Sediment type distribution in Chesapeake Bay. Image adapted from *Surficial Sediments of Chesapeake Bay, Maryland: Physical Characteristics and Sediment Budget* Kerhin, R. T. et al., 1988. Maryland Geological Survey.

Research Various studies conducted over three decades have explored the causes of the disappearance of oysters and submersed grasses. The decline in oyster numbers has been linked to overfishing, pollution and disease, while the disappearance of aquatic grasses has been linked to light reductions from eutrophication and turbidity (Dennison et al., 1993; Ernst, 2003). Associated research has also elucidated the critical role that filtration by oysters played in the ecology of the bay, including removing phytoplankton and particulate matter from the water column and enhancing denitrification in shallow water (Newell, 1988). Similarly, aquatic grass beds played a key role in reducing water movement and hence trapping and binding of suspended sediments (Fonseca et al., 1982). Continuing research is assessing all aspects of habitat requirements for potential re-establishment of aquatic grass in the bay (Koch, 2001).

Working toward a solution The elucidation of the key ecological roles of oysters and aquatic grasses has led to oyster restocking and seagrass restoration programs (Table 2). Large scale oyster hatcheries and aquatic grass nurseries have been built, and replanting programs developed. These have been partially supported by volunteer networks. Introduction of an exotic oyster species with disease resistance is being contemplated. There are significant caveats associated with these restoration programs: degraded water quality and/or degraded habitat reduce the likelihood of survival of oyster spat and aquatic grass propagules. Another difficulty is that direct restoration is typically both very expensive and highly labor intensive.

1.10. Chesapeake Bay Sedimentation

Problem Chesapeake Bay is a natural sedimentation basin, and on a geologic time scale is steadily filling in (Fig. 7b). Rates of sedimentation have been accelerated due to land use changes (e.g. increasing agriculture) leading to runoff and deposition, as well as sea level rise and resultant coastal erosion (Table 2). The major Susquehanna River dam (Conowingo dam) is rapidly filling with sediment and is predicted to reach nutrient and sediment capacity in less than 15 years (Langland, 1998). Further upstream on the Susquehanna River, the Safe Harbor and Holtwood Dams have already reached their capacity for storing nutrients and sediment (Langland, 1998). Once the Conowingo dam is full, the mean annual sediment load to Chesapeake Bay will increase by 250% and the phosphorus load by 70% (Langland & Hainly, 1997). The deep water approaches to Baltimore Harbor currently require annual dredging to remain navigable. Currently, the dredge spoil from this navigation dredging is being relocated to a couple of artificially constructed islands in the bay.

Research The role of sedimentation in the bay has been elucidated through a variety of research approaches. It is now realized that the interplay between relative sea level rise and shoreline processes has a large influence upon sediment dynamics. Chesapeake Bay is a region of rapid relative sea level rise (30 cm in the last 100 years, nearly twice the global average), contributing to extensive salt marsh erosion, with several low islands already drowned (Stevenson & Kearney, 1996). Groundwater extraction, principally for agriculture, results in land subsidence and therefore further accelerates relative sea level rise (Davis, 1985).

Working toward a solution Various incentive schemes have been devised for farmers to maintain cover crops during the period of the year when the fields are fallow, to aid in reducing erosion and sediment deposition to the bay (Table 2). A scheme for using dredge spoil to augment the marshes and islands that are eroding is being explored. This could alleviate the problem of dredge disposal as well as preserving salt marsh habitat.

1.11. Chesapeake Bay Harmful Algal Blooms

Problem Harmful algal blooms in Chesapeake Bay are increasing in frequency and diversity (Glibert & Magnien, in press) (Table 2). This has been related to increased nutrients entering the estuary, the major sources being fertilizer, manure, atmospheric deposition and sewage (Glibert & Magnien, in press). There are two main detrimental effects of these blooms; firstly, the direct effect of the toxins upon humans and bay fauna and secondly, hypoxia or even anoxia in bay waters (Table 3). Species such as *Pfiesteria piscicida*, *Microcystis aeruginosa* and *Dinophysis* sp. have been linked to human health issues, with blooms resulting in river closures. The anoxia associated with large decaying blooms of *Prorocentrum minimum* have caused fish and shellfish stress and death (Table 3; Glibert & Magnien, in press).

Research The harmful algal bloom species that occur in Chesapeake Bay are very diverse, from the dinoflagellate *P. piscicida* with a complex life cycle to the nitrogen fixing cyanobacteria *M. aeruginosa*. As a result the specific triggers of blooms can be more complex than simply the presence of higher nutrient concentrations. As well as diverse research into physiology, grazing, population dynamics and triggers to toxin production, continued research employing continuous in situ nutrient sensors is helping to elucidate the interactions of nutrient pulses and salinity as well as the effects of nutrient ratios (nitrogen: phosphorus: silicon) and how these balances may control both the species and intensity of harmful algal blooms (Table 3) (Glibert & Magnien, in press).

Working toward a solution Harmful algal blooms have implications for political, medical and scientific communities, therefore the solution must encompass all these groups. To this end, a task force has been established between State of Maryland agencies and the medical community to address

Table 3: Common Harmful Algal Bloom (HAB) species in Chesapeake Bay, with major impacts and causes.

Algal species	**Toxic effects**	**Causes hypoxia**	**Probable bloom cause**
Pfiesteria piscicida	Yes	No	Nutrients, other algal blooms
Prorocentrum minimum	No (?)	Yes	Freshwater, nutrients
Microcystis aeruginosa	Yes	No	Nutrients (phosphorus)
Dinophysis sp.	(?)	No	High salinity, nutrients

After Glibert & Magnien (in press).

specific issues and recommend river closures where required. In the Potomac River, phosphate removal from sewage has been effective in reducing 70 $\mu g\ l^{-1}$ blooms of *Microcystis* spp to populations generally less than 20 $\mu g\ l^{-1}$ (Anderson et al., 2002). The general issue of nutrient inputs to Chesapeake Bay is a continuing problem. However, in areas where point sources were the primary concern, some reductions in nutrient loads have been achieved with resultant reduction in harmful algal blooms in these areas (Anderson et al., 2002; Glibert & Magnien, in press).

1.12. Overcoming Challenges

There are political, financial and scientific challenges to be overcome in addressing environmental problems. Drawing upon case studies presented at the International Riverfestival held in Brisbane (Australia) over the past several years, examples of programs that have overcome some of these challenges will be discussed. Only when all three aspects of the challenges have been addressed can a solution be achieved and in every separate case the relative difficulty of the three aspects of the environmental problem varied due to historical and site specific factors.

The physical size of the system also influences the difficulty of the political, financial and scientific challenges. The challenges in a larger system (such as Chesapeake Bay) will often have a greater complexity of challenge in finding environmental solutions, simply due to the watershed and airshed having a greater diversity of anthropogenic activity within its boundaries. The Mersey Basin (United Kingdom) is relatively small and defined, so the scientific issues were relatively simple-whereas in the Mekong River (SE Asia), the massive size of the system meant that the political challenges just to coordinate a management effort were enormous.

1.13. Healthy Waterways Campaign Overcomes Population Growth

The challenge that *population growth counteracts any progress made with management interventions* is one that most coastal watersheds face as both population growth and coastal migration have led to increased human population impacts in coastal regions. Moreton Bay is in the region of largest population growth in Australia, so the primary issue in working towards a solution for environmental problems was political. The response to this challenge was a proactive program which accounted for population growth and new development. The result was a plan to manage population growth while maintaining ecosystem health. The Healthy Waterways Campaign in south-east Queensland, Australia, has achieved water quality improvements in spite of being the fastest growing urban area of Australia, and one of the fastest growing areas of the world (Abal et al., 2001). A key aspect of this program is that it incorporates the entire watershed, and largely encompasses the ecological footprint of the city of Brisbane (water storages, farm land, recreational areas, suburban and urban regions). As a result, population growth and its consequences in terms of services required throughout the region are considered in working towards solving the environmental problems (http://www.healthywaterways.env.qld.gov.au).

1.14. Chesapeake Bay Blues

Chesapeake Bay is the biggest estuary in the USA and significant progress has been made in terms of political (well established management), financial (well supported) and biological (well studied) aspects of solving the environmental problems, however, the Bay is still not improving. The term *Chesapeake Bay Blues* was coined by one of three recent publications, all recognizing that despite 20 years of active effort to improve the health of Chesapeake Bay, the biggest issues of nutrient loading, habitat loss, increased sedimentation and harmful algal blooms remain or have worsened (Boesch & Greer, 2003; Ernst, 2003; Horton, 2003). The continued research and management of Chesapeake Bay over the past 30 years has seen an enormous increase in the understanding of processes within the bay. Small regions have shown some positive signs resulting from reduced sewage inputs (Boynton et al., 1996). Other improvements include increases in striped bass and Atlantic croaker populations. Overall, however, Chesapeake Bay is languishing in a state which has large anoxic events, oyster and seagrass decline, coastal erosion, fish kills and water unsafe for swimming (Ernst, 2003). As early as the 1930s the key issues of sewage inputs, over-harvesting and sediment inputs had been identified, but state boundaries and political forces have hindered significant improvement within the bay (Ernst, 2003). If recent trends continue, it is expected that nutrient loads will increase and air quality will degrade, forest cover will decline, residential sprawl will continue to expand and aquatic grasses and fisheries will decline further (Boesch & Greer, 2003). To move forward, stronger links between political, financial and scientific elements are required. A better balance between research, management and monitoring, focused on effective feedback, will be required to implement established goals and solve the environmental challenges currently facing the bay (Orth et al., 2002; Boesch & Greer, 2003; Dennison, in press; Glibert & Magnien, in press).

In Moreton Bay, the political issues of planning for increased population growth have been able to generate funds and target scientific understanding which has led to significant gains in solving the currently known environmental problems in the bay. In Chesapeake Bay, significant gains in management structure and goal setting, funding and scientific understanding have not resulted in significant gains to the main environmental problems currently known. The Chesapeake Bay community now has the resources to move forward on the often difficult problems such as diffuse agricultural and atmospheric nutrient inputs. If effective, this process can be an example for coastal systems throughout the world. Different systems are at different stages of the process of solving their environmental problems. One exemplary case is the Mersey Basin in the United Kingdom, which solved their issues of point source nutrient pollution, while the Mekong River is succeeding in overcoming significant political issues to establish a management framework for their restoration efforts (http://www.chesapeakebay.net).

1.15. Mersey Basin Campaign Overcomes Cost Considerations

The challenge that *It will cost too much* is one that practitioners around the world are facing, in the Mersey Basin the political solution was present and the biological solution was relatively simple, excessive point source nutrient inputs. The issue of funding can be manifested in a variety of ways, but often represents the single largest challenge. The appropriate response to this challenge is that investments in protection and restoration are cheaper now than they will ever be in the future, and these investments can stimulate local economies. The Mersey Basin Campaign is a 25 year campaign in the world's first industrialized region of NW England. The Mersey basin has two large cities, Liverpool and Manchester, with six million inhabitants. The Mersey River was highly polluted with raw sewage discharges as recently as the 1980s. The land values along the river's edge were actually negative-local government was unsuccessful in giving land away to developers even with a cash incentive. However, a concerted clean-up effort has made the water safe for swimming, and the negative land values are now positive, including the development of a five star hotel on the banks of the river. The Mersey Basin Campaign was the first recipient of the International Riverprize in 1999 (http://www.merseybasin.org.uk).

1.16. Mekong River Commission Overcomes Jurisdictional Issues

A significant initial stage in reaching a solution to environmental problems is to overcome diverse political requirements and priorities. The challenge that *There* *are too many different jurisdictions and stakeholders with divergent views* is one that most regional scale programs face. Coastal watersheds tend to cross jurisdictional boundaries and there are certain to be stakeholders with divergent views in any region. The response to this challenge is that a participatory process can create a shared vision among a variety of stakeholders. The Mekong River Commission has put together a multi-national program involving Cambodia, Lao PDR, Thailand and Vietnam, with links to China. These are countries that have been involved in bitter conflicts with each other within this present generation. The Mekong is the 8th largest river system (in water volume) globally and supports major fisheries. There are 17 million people and 70 ethnic minorities in the Mekong watershed. The Mekong River Commission was the 2002 recipient of the International Riverprize, based on their proven ability to develop a participatory process among their stakeholders (http://www.mrcmekong.org/).

1.17. Conclusions

In many ways, these coastal management programs can be viewed as ‘experiments’. While not having replication or controls, the challenge for science practitioners is to apply rigor to these coastal management case studies to develop global standards and work towards more effective management practices. The global trend for increasing human pressures on coastal regions has created a suite of difficult, but tractable environmental problems. Obtaining sustainable solutions to important environmental problems needs to be a major scientific focus of the next half century. The presented case studies of coastal ecosystem management programs serve to introduce environmental problems that are not unique to one region and demonstrate that solutions are possible for a wide variety of problems.

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This chapter represents outcomes from a series of discussions over the past several years with various different groups of scientists. The case study examples touch on research results from a plethora of scientists. The attempt here will be to acknowledge the programs and groups, and cite some of the key individuals but not provide a comprehensive listing. The Moreton Bay case study was developed as part of the Healthy Waterways Campaign team, in particular Eva Abal, Paul Greenfield and colleagues in the University of Queensland Marine Botany group. The Chesapeake Bay case study was developed with colleagues at the University of Maryland Center for Environmental Science (UMCES), in particular, Donald Boesch, Walter Boynton and Dave Nemazie. Assistance with graphics and editing was provided by Tracey Saxby, Adrian Jones, Ben Longstaff and Chris Roelfsema. Support from the Integration and Application Network, UMCES is acknowledged. UMCES contribution number 3753.

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Chapter 2

Wetland Utilization and Protection in China

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Abstract. Wetland is an important component of the global ecosystem and plays a key role in water conservation, environmental clean-up and biodiversity conservation. China possesses approximately 10% of the world's total wetlands. However, with increasing population and economic growth, wetlands have been over-exploited, which resulted in a sharp decrease in quantity, quality deterioration and degradation of ecological functions. In this chapter, concrete measures for protection of wetland resources are proposed. Utilization of coastal wetland in Jiangsu province exemplified the prospects of wetland utilization and protection, and may serve as an example for wetland utilization and protection in other parts of China.

2.1. Brief Introduction to Wetland Resources in China

As an indispensable component of the global ecosystem, wetlands are among the most important natural habitats that support biodiversity and provide subsistence for humans. Although wetlands occupy only 6% of the earth's surface, they support roughly 20% of all living organisms on this planet.

China has 65.94×10^6 ha of wetlands (excluding rivers and ponds), accounting for about 10% of the world's total. In terms of wetland area, China ranks the first in Asia and the fourth in the world. According to a survey, China's natural wetlands amount to 25.94×10^6 ha including 11.97×10^6 ha marshes, 9.10×10^6 ha natural lakes, 2.17×10^6 ha inter-tidal flats and 2.70×10^6 ha shallow seas (less than 5 m deep at low tide) (State Forestry Administration, 2000). By 2001, China had 21 wetlands of international importance (Fig. 1).

Wetlands are widely distributed, unique ecosystems of various types and with complex structures. Their ecological functions include water conservation, environmental clean-up, biodiversity conservation and biomass production, etc.

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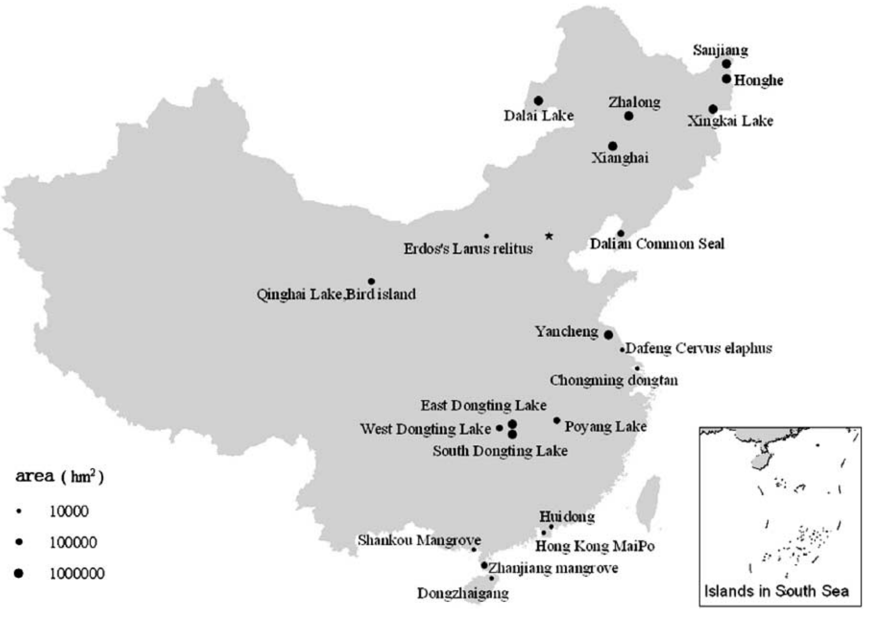

Figure 1: Chinese wetlands of international importance.

With increasing demand for material and expansion of territories, human intervention and utilization of wetlands also intensified. Therefore, wise utilization and protection of China's wetland resources is of great significance for the ecological balance of China's natural resources, environmental protection and sustainable development of the national economy.

2.2. Problems Arising from Wetland Exploitation and Utilization

Over-exploitation of wetlands and inappropriate farming practices have caused a series of problems such as reduction in the quantity of wetlands, quality deterioration and degradation of their ecological functions and reduced biodiversity, etc.

2.2.1. Over-Exploitation and Shrinking Wetlands

Wetland shrinkage in China can be mainly ascribed to wetland reclamation and urbanization. This is especially true in the densely populated coastal areas and

Table 1: Temporal changes (in km^2) of lakes in the middle stream of Yangtze River.

Period	Dongting Lake	Jinghan Lakes	Four Lakes region
The 1920s–1930s	4,206	8,330	
The 1950s	4,009	5,960	2,030
The 1970s	2,507.87	2,373	
The 1980s	2,146.9	2,983	844
The 1990s	1,502.7	2,608	707.34

Source: Yu (1999).

lakesides. Wetlands have been disappearing at a rate of 20,000 ha per year as a consequence of conversion of lakeshore to farmland. On Sanjiang (the Three Rivers) Plain, where wetlands are widely distributed, farmland increased by 4.6 times from 786,000 ha in 1949 to 3.668×10^6 ha in 1995 (Zhang et al., 2001). The number of lakes (larger than 0.5 km^2) in the middle and lower streams of the Yangtze River was reduced by 43.5% over 30 years from 1950s to 1980s (Table 1).

Mudflat was reclaimed and thus shrank more than any other wetlands. Between the 1950s and the 1980s, 2×10^6 ha of mudflat had been reclaimed, accounting for 50% of the total in China. As a consequence of aquaculture and tideland reclamation, the mangrove area in the whole country was reduced by 50% from 40,000 ha in 1957 to 18,000 ha in 1986.

2.2.2. Wetlands' Quality Deterioration

Based on past experience and current status of wetland degradation in key regions and in the whole country, the main influencing factors of wetland degradation can be categorized as follows: wetland reclamation and exploitation, improper use of biological resources, wetland pollution, utilization of wetland water resources and water engineering projects, sedimentation, coastal erosion as well as urbanization and tourism. Different influencing factors pertained in different regions and had different consequences and trends (see Table 2).

Wetland reclamation is now strictly prohibited. However, other factors are still exerting their effects on wetland degradation. Therefore, the trend of wetland degradation is unlikely to be altered in the near future.

Table 2: Influencing factors of wetland degradation and their trends.

Influencing factors	**Affected regions**	**Consequences**	**Trends**
Reclamation and over-exploitation of wetland resources	Densely populated coastal area and lakeshore	Wetland shrinkage and degradation of ecological functions	Reclamation for farming before the 1980s and for aquaculture between the 1980s and the 1990s, the impact will be reduced in the future
Improper use of wetland biological resources	Excessive fishing in lakes, reservoirs and coastal area, mangrove shrinkage in the coastal area	Reduced biodiversity, degradation of ecological functions and habitats destroyed	Productivity reduced significantly, but huge demand still remains
Wetland pollution	Nearly all wetlands are affected by anthropogenic activities, especially in developed areas	Deteriorating water quality, increasing pollutants' concentration, loss of water decontamination function, reduced biodiversity	Serious situation at present, and impact will be intensified
Improper use of wetland water resources and water engineering projects	Northwestern and Northern China, hydro-engineering projects on big rivers	Water sources shrank or dried-up, habitats destroyed, degradation of ecological functions	Impact will last for a long period, deteriorating in Northwestern and Northern China

Sedimentation	Nationwide, especially in Central and Eastern China	Wetland shrinkage, loss of water conservation function	Deteriorating
Coastal erosion	Coastal area, especially Southeastern China	Mudflat shrinkage	Impact still remains
Urbanization and tourism	Southeastern coastal area including Yangtze and Pearl River Deltas	Wetlands demolished or isolated, habitats of water birds changed	More impact by urbanization and increasing impact by tourism

Source: Zhang et al. (2001).

2.2.3. Ecological Degradation and Reduced Biodiversity

According to statistics, higher plants of 172 families, 495 genera and 1,642 species (including varieties) can be found in China's wetlands. They account for 48.7, 15.5 and 5.5 of the total higher plant families, genera and species in China, respectively. Microbial biomass in marshes can be as high as 7.26×10^7 cfu/g DW, of which bacteria, actinomycosis and fungi constitute 99.62, 0.032 and 0.34%, respectively. About 68 soil animals from 5 kingdoms, 10 classes, 2 orders and 37 families can be found in swamps (Tian et al., 2002). Due to human intervention, natural habitats of some wetland flora or fauna were destroyed. As a consequence, composition, structure and quantity of wetland biological communities were changed. In worse scenarios, the number of certain wetland species will decrease sharply or even die out. Simplification and reorganization of the wetland biological community, changes in dominant species or even converse evolution may take place, which eventually leads to reduced or even loss of biodiversity.

2.3. Measures for Protection of China's Wetlands

The Chinese government has attached great importance to the eco-environment development. A wetland recovery plan has been implemented in the middle and lower streams of the Yangtze River. The trend of human-induced wetland shrinkage is contained. However, mudflat exploitation has not been brought under control. With establishment of more wetland nature reserves and strengthening of wetland management, wetland protection will be reinforced. But, as a whole, the trend of wetland degradation is unlikely to be reversed in the near future.

The following measures can be taken to reinforce management and protection of wetland resources.

1. Attention should be given to both protection and utilization. The protection and utilization of wetland resources should be considered from the viewpoint of sustainable development. Economic, social and ecological benefits should all be taken into consideration. Based on a better understanding of natural conditions, ecological functions of the wetland and demands from society, the best utilization pattern (farming, forestry, animal husbandry or fishery, etc.) can then be decided. In the meantime, principals of landscape ecology and agro-ecology should be applied to establish integrated wetland utilization patterns that are based on modern, highly efficient, agro-ecological methods.
2. Wetland protection lawmaking and implementation should be reinforced.
3. Further studies on wetland resources are greatly needed, which include wetland structure and functions, wise utilization and protection of wetland resources,

restoration of degraded wetland ecosystems, wetland ecosystem indicator systems, wetland ecology and health diagnoses, and wetland contribution to globe warming, etc.

2.4. Utilization and Protection of Coastal Wetland in Jiangsu Province

The coastline of Jiangsu province is 1,000 km long. There are 650,000 ha of coastal wetland (mainly mudflat) in Jiangsu province, which accounts for one-quarter of the total mudflat in China. Mudflat is increasing at a rate of about 1,300 ha each year (Yang et al., 1997). The potential use of the mudflat can be summarized as follows:

(1) Cultivation and amelioration of enclosed mudflat. Yields of cereal crops and cotton can reach at least 9 and 0.75 t per ha each year.
(2) Amelioration of cultivated mudflat with medium and low productivity. By means of increasing input, change of product composition and scientific management, yields of cereal crops and cotton can increase by 2.25 and 0.75 t per ha. Yields of aquatic products are expected to be doubled.
(3) Enclosing tideland for cultivation, as well as for integrated exploitation and utilization of mudflat. About 33,300 ha of mudflat in Jiangsu province can be enclosed in the near future.
(4) Intensified processing of cereal, cotton, forest, fruit, grass, livestock and aquatic products also has considerable potential.

2.4.1. Problems Associated with Utilization of Mudflat

Problems associated with utilization of mudflat can be summarized as follows:

(a) irrigation and water conservancy facilities need to be improved;
(b) where there is freshwater shortage, groundwater should be exploited wisely;
(c) low and unbalanced nutrients in the soil;
(d) the scale and efficiency of farming, animal husbandry and aquaculture need to be improved; and
(e) more financial support and R&D (research and development) are needed for mudflat exploitation.

To deal with the above-mentioned issues, monetary as well as scientific and technological investments are needed in mudflat exploitation and protection. Integrated exploitation of mudflat should be extended fully. Mudflat exploitation can be intensified in four aspects: (1) improving productivity and benefits of land with mediocre and low yields; (2) cultivation of enclosed mudflat; (3) integrated utilization of tideland; and (4) enclosing more tideland for cultivation. By doing so, the regional agricultural potential of mudflat in Jiangsu province can be fully exploited. This will help upgrade natural resources' utilization and increase benefits, thus facilitating regional economic development. In the meantime, this will provide an example for exploitation and utilization of mudflat resources in other parts of China.

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Chapter 3

Ecological and Environmental Function of Wetland Landscape in the Liaohe Delta

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Abstract. This chapter summarized the ecological functions of the natural wetland in the Liaohe Delta. The annual surface runoff amounted to $8,298 \times 10^6$ m^3. The evapotranspiration of the reed field was 37.5% higher than that of the large water body during growth season. The 95% water replacement rate was calculated as 11 days of continuous rainfall with 912 mm of precipitation. The aboveground bio-productivity of the reed field was about 14 t/ha in average. Nearly 60% of TN and 50% of TP can be removed by the reed system from paper factory effluents. The seasonal dynamics of CH_4 emission was positively related to the temperature, which was higher in summer and lower in winter. The extensive natural wetland in the Liaohe Delta is of great importance in bio-conservation because of the large number of wild lives residing and migrating through this place, including a number of rare species.

3.1. Introduction

The littoral wetlands are transitional areas between the continent and the sea. These areas belong to a vulnerable ecotone, making them important for nature preservation and as a buffer zone against global sea level rising.

The Liaohe Delta is located within the range of 121°35′–122°55′E and 40°40′–41°25′N, with an area of about 4,000 km^2. Several large rivers run into the sea here with 11.7 billion m^3 of water every year. Counteracted by sea tides, 76 million t of sedimentation is accumulated in the delta annually. The reed marsh constitutes the main part of the delta, with an area of 1,000 km^2, which is the largest reed field in the world. The climate of the research area is temperate monsoon, with an annual temperature of 8.3 °C, and an annual precipitation of 611.6 mm. More than 70% of the rainfall is in summer, with high evaporation and natural disasters such as draught, waterlogging, windstorm, hail and storm tide. Wetlands in the Liaohe

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delta are mainly seasonally waterlogged (64% of the total), including paddy field (58%) and reed marsh (32.8%) (Xiao, 1994).

The Liaohe Delta is located on the transition zone between the Bohai Sea and the dry land, at the convergence of fresh and salty water, affected by both the sea and island. Complex driving mechanisms formed the many and various natural wetlands and ecological environments, such as river wetland, estuary wetland, swampland, meadow wetland, and coastal mudflat wetland. There were also artificial wetlands including reservoir, paddy field and man-made salt marsh that are influenced by intensive human activities.

Taking the Liaohe Delta as the study area, this chapter summarized the ecological and environmental function of wetlands including water regulation, biomass production, biodiversity production, biological nutrient circulation, nutrient reduction and greenhouse gas emission.

3.2. Hydrological Adjustment of Wetland

These wetlands are playing important roles in hydrological adjustment, i.e. regulating the water storage, flooding, flush plow, and surface and ground water exchange by irrigation, discharge and evapotranspiration (Xiao et al., 2001).

3.2.1. Wetland Water Storage Capacity and Reed Field Evapotranspiration

The potential ground surface impoundment of the delta region includes the maximal runoffs of rivers, maximum impoundments of reservoirs, reed fields, salt pans, shrimp ponds, paddy fields, and high-flow year measurement or calculated storage cubage of ponds and canals. Two rivers run through the Liaohe Delta, the Shuangtaizi River and the Raoyang River, with lengths of 116 and 71 km, respectively, and a total storage capacity of 209.3×10^6 m^3. There are seven plain reservoirs in the Delta, with a total storage capacity of 139×10^6 m^3. The storage capacities of reed field, paddy field, and canals and ponds are 800×10^6 m^3, 237×10^6 m^3 and 366×10^6 m^3, respectively. The sum total of the above values provides an estimate for the potential ground surface impoundment of the delta of $1{,}763 \times 10^6$ m^3. The total water resources in the Liaohe Delta is $8{,}298 \times 10^6$ m^3, among which, the annual river runoff is $7{,}204 \times 10^6$ m^3, taking 86.8% of the total. The depth of the ground surface runoff is 78.3 mm and the annual surface runoff amount is 258×10^6 m^3, taking 3.1% of the total. The exploitable underground fresh water resource is 836×10^6 m^3, representing 10.1% of the total (Xiao, 1994).

Table 1: Comparison of evaporations between large water body and reed marsh in Panjin city (mm).

Year	**Large water body**								**Reed field**
	1997		**1998**		**1999**		**Mean**		
Month	**φ20 cm**	E_{601}	**φ20 cm**	E_{601}	**φ20 cm**	E_{601}	**φ20 cm**	E_{601}	
June	245.0	149.5	169.6	103.5	221.5	135.1	212.0	129.4	174.0
July	212.0	125.1	142.4	84.0	206.5	121.8	187.0	110.3	213.9
August	164.7	100.5	168.6	102.8	193.5	118.0	175.6	107.1	161.2
September	154.3	94.1	169.1	103.2	183.9	112.2	169.1	103.2	87.0
October	127.2	76.3	132.3	79.4	120.4	72.2	126.6	76.0	86.8
June–Oct.	903.2	545.5	782.0	472.9	925.8	559.3	870.3	525.9	722.9

Note: φ20 cm and E_{601} are different standard plates for measuring large water body evaporation under natural condition.

The mean annual precipitation of the research area is about 620–640 mm, and the observed value for annual evaporation from a small water body was 1,636–1,656 mm. By conversion, the year-round water surface evaporation is 933–941 mm, with 371.3 and 301.5 mm of the evaporation in summer and spring, respectively. The dry land annual evaporation was calculated as 541–555 mm.

According to the field observations during 1997–1999 in the Yangjuanzi reed farm in Panjin city, the daily average evapotranspiration from May to October was 4.6, 5.8, 6.9, 5.2, 2.9, and 2.8 mm, respectively. The highest evapotranspiration occurred in July due to the high temperature and the fastest reed growth. Comparisons between the evapotranspiration of a reed field and a large water body are shown in Table 1.

In Table 1, we see that the evapotranspiration from the reed field (including water surface evaporation, and reed transpiration) during the growing season from June to October was 722.9 mm, about 37.5% higher than evaporation from the large water body (E_{601}) (525.9 mm). This was due to high rate of plant transpiration in the reed field.

3.2.2. Water Replacement Rate

Water replacement rate is a criterion of wetland openness that directly affects the wetland chemical and biological processes. Water replacement rate represents the speed of water renewal in the wetland landscape, whose reciprocal is the retention time of water in wetlands. It can be calculated as:

$$\beta = Q_i/V$$

Where β is the replacement rate; Q_i is the rate of flow through the wetlands; and V is the impoundment capacity of the wetlands. Suppose the water retention time is t, rate of inflow $Q_i = V/t$, the first day impoundment is $(1 + 1/t)V$, and the impoundment of the nth day is $(1 + n/t)V$. If the water quality concentration in the wetlands is a and that of the inflow water is b, the water quality concentration of the wetlands after n days is going to be changed to $(ta + b)/(t + n)$.

Suppose the wetland impoundment has reached the capacity C, i.e. $V = C$, then the average volume of water running through the wetland per day is C/t.

On the first day, the water will decrease by $1/t$, leaving $(1 - 1/t)C$ until the nth day, when only $(1 - 1/t)^n C$ is left. If the replacement is defined as finished when the remaining water is 5% (or 1%), we can assign a value to t, and calculate n. For example, let $t = 10$, i.e. the water running through the wetlands is 1/10 of the impoundment, then 29 days later 5% water remains, and approximately 44 days later less than 1% water is left (Table 2).

We calculated the water replacement for the wetlands in the Liaohe delta on August 21st, 1997 as an example. The precipitation on August 20th was 92.7 mm, and the soil was saturated already after the rain. On August 21st another rainfall of 114 mm started. The runoff generated at the apoapsis of the region to the estuary took about 24 h. Therefore

Q_i = rainfall intensity × area $= 0.114\text{m} \times 3{,}959 \times 10^6\text{m}^2 = 4.5 \times 10^8(\text{m}^3/\text{day})$

$C = 1.9 \times 10^9\text{m}^3$; $Q_i/C = 1 - 0.24$.

Let $(1 - 0.24)^n < 0.05$ then, $n = 11$ day.

Since soil seepage rate is lower than 5% of rainfall intensity, it can be ignored. According to the calculation, we know that 11 days of continuous rainfall and

Table 2: Duration of wetland water replacement (days).

Water replacement rate	**Impoundment/C**			
	<5%	**<1%**	**<0.5%**	**<0.1%**
0.1	29	44	51	66
0.2	14	21	24	31
0.3	9	13	15	20
0.4	6	9	11	14
0.5	5	7	8	10
0.6	4	6	6	8
0.7	3	4	5	6
0.8	2	3	4	5
0.9	2	3	3	4

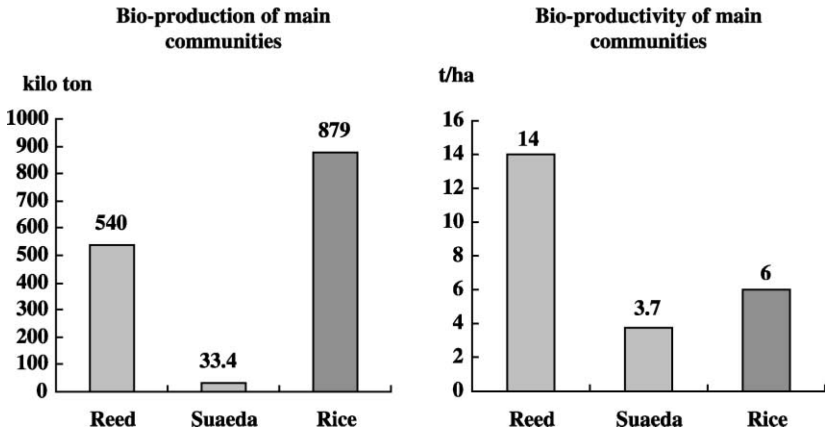

Figure 1: Bio-production and productivity of the main communities in the Liaohe delta. The reed and *Suaeda* figures are for above ground biomass, while the rice is grain production.

a precipitation of 912 mm, is required in order to replace 95% of the water in the wetlands in the delta.

3.3. Biomass Production and Output

The Liaohe delta has three main wetland vegetation types: reed, *Suaeda* and rice. The production and productivity of these systems are shown in Fig. 1. In 1998, the average bio-productivity of the reed community reached 14 t/ha/yr, and the total production was around 540,000 t. About 92% of the reed is cut annually in winter and used as raw material in the nearby paper factories. The biomass productivity of the *Suaeda* community is about 3.7 t/ha/yr (Li, 2000). And the productivity of rice has been as high as 6 t/ha/yr (grain only) in the recent years.

The high productivity of the main ecosystems in the wetland of the Liaohe Delta provides material basis for other functions such as water regulation, soil formation, bio-protection and purification.

3.4. Purification in the Wetland

3.4.1. Waste Water Irrigation in the Reed Field

By irrigating the reed with wastewater, both ecological and economical benefits can be obtained in the local area (Li, 2000). Wastewater can increase reed

Table 3: Comparison between wastewater and normal water irrigated reed growth.

Year	Irrigated by	Height (cm)	Stem diameter (cm)	Number of gnarls	Productivity (t/ha)	Production increase (%)
1981	Waste water	320	0.80	22	10.6	26.3
	Normal water	270	0.70	18	8.6	–
1982	Waste water	310	0.85	21	10.4	16.8
	Normal water	275	0.75	20	8.9	–
1983	Waste water	337	0.75	24	15.0	26.0
	Normal water	297	0.70	22	11.0	–

Song and Sun (1984).

productivity better than ordinary water, because it contains more nutrient elements. Also, it can partly solve the water shortage problem in spring and avoid coastal seawater pollution. This is why wastewater irrigation has been encouraged in the recent decades.

Increase More Production Than Normal Water Irrigation. Results of a field experiment carried out in the 1980s showed that the height of reed irrigated with waste water was 45–50 cm higher than that irrigated with normal water (Song and Sun, 1984), while the productivity was about 17–26% higher. In the field, the reed irrigated with wastewater had strong stems and dark green leaves, with an optimum stem density (Table 3).

Amelioration of the Water Shortage Problem in Spring. Spring (March–May) is usually quite dry in the Liaohe Delta. The evaporation is 17 times higher than the precipitation in March, while the highest evaporation occurs in May (281.5 mm). The total spring precipitation in the Liaohe Delta is only 96.5 mm, about 15.5% of the annual rainfall. These figures are far lower than is required for natural vegetation growth, not to say supplying the local industrial and agricultural needs. If the reed fields are irrigated with waste water from upstream factories, no heavy damage will be done to the reed growth, while the water with better quality can be saved for agriculture and industry.

Prevention of Coastal Water Pollution. If the wastewater with a high pollutant concentration is discharged directly into shallow sea, in combination with suitable conditions for some algae species, it can cause great problems of algal growth. The main pollutants in the seawater are inorganic nitrogen, inorganic phosphorous,

Table 4: Runoff, COD and nitrogen release into the Liaodong Bay of the main rivers (Li, 2000).

River name	Runoff ($\times 10^9$ m^3/yr)	COD (t/yr)	NH_4^+–N (mg/l)	NO_2–N (mg/l)	NO_3–N (mg/l)
Shuangtai R.	2.1	21,697	0.37–4.23	0.002–0.040	0.11–1.40
Daliao R.	4.0	53,903	2.30–104.00	0.000–0.088	0.00–2.38
Daling R.	2.0	49432			
Xiaoling R.	0.4	2,245			

and oil. In the Bohai sea, the concentration of nitrogen above the National Standard increased from 16% in 1996 to 68% in 1997 (Li, 2000). The problem is especially serious near large river mouths and large cities. For example, in July 1991, the large "red tide" (algae blooming) in Liaodong Bay covered thousands of square kilometres of the sea surface. This caused great damage to both the local fishing and the shrimp/crab breeding industries.

According to the data released from the local environment monitoring organization, approximately 130,000 t of carbon oxygen demand (COD) per year is brought directly into the sea by the major rivers flowing into Liaodong Bay, together with other nutrient elements such as nitrogen and phosphorous (Table 4).

Improving Soil Fertility. Wastewater usually contains more nutrients than ordinary river water. After several years of irrigation with wastewater, peat soil in the reed marsh often becomes more fertile. No production decrease has been observed, although a large amount of dry material is removed each year as raw material for paper factories. So far no accumulation problem has been observed after more than 30 years of wastewater irrigation.

3.4.2. Purification of the Reed Field to Waste Water from Paper Factory

During 1997–1998, intensive field experiments in the Liaohe Delta were made to investigate the purification function of reed marsh and the canal system. The field was irrigated with wastewater upstream from paper factories, three times a year in spring. In the reed field, ground water was sampled 3 days after irrigation at 0, 40, 60 and 80 centimetre depths with the Lysimeter system. Samples were acidified and analysed with standard methods within 3 days after sampling. The results are given in Table 5.

It is clear in Table 5 that the value for COD, TN and TP decreased downwards in the profile. The reduction rate for organic nitrogen was especially high

Table 5: Purification of the reed marsh system to waste water from paper factory.

Depth	**COD (mg/L)**	**TN (mg/L)**	**Organ−N (mg/L)**	**NH_4^+−N (mg/L)**	**NO_3^-−N (mg/L)**	**NO_2^-−N (mg/L)**	**TP (mg/L)**	**SRP (mg/L)**
0 cm	82.69	3.129	2.676	1.36	0.069	ND	0.150	0.043
40 cm	69.63	1.922	1.212	1.33	0.142	ND	0.082	0.047
60 cm	73.98	1.204	0.861	1.23	0.054	ND	0.067	0.024
80 cm	60.93	1.255	0.811	1.32	0.105	ND	0.080	0.028
Rdc (%)	26.3	59.9	69.7	2.9	–	–	46.7	35.4

Concentration values were averaged from 2 years of observation; ND, not detected. Rdc is the reduction rate, calculated as the difference between pollutant concentration in the surface water and 80 cm groundwater divided by the surface concentration value; Li et al. (1999).

(about 70%), because of absorption by the rhizome system. Peat soil was also highly absorptive.

The experiment in the reed field was mainly to measure the vertical retention rate of the reed–soil system for polluted water. In addition, the horizontal subsurface flow in the rhizosphere also has a high reduction rate to some nutrients like nitrogen and phosphorous (Yin and Lan, 1995). In combination, the reduction rate of the reed–soil system for nutritious elements is very high. Thus, the pollution content of ground water discharging from the reed marsh system into the sea is greatly reduced.

3.5. Methane (CH_4) Emission from the Natural Wetland

Greenhouse gas emission has become a global problem in the recent decades, on account of its great potential effect on climate change. Methane (CH_4) is considered as the second most serious greenhouse gas after CO_2, and worldwide wetlands emit about 55–150 Tg/yr. The following account is based on a field experiment in the natural reed marsh of the Liaohe Delta (Huang et al., 2001). The high reed stems (>2 m) and deep water (often > 30 cm) made monitoring very difficult.

3.5.1. Seasonal Dynamics of CH_4 Emission

CH_4 emission from the wetland is a combination of CH_4 production, re-oxidization and transportation. Fig. 2 demonstrates the seasonal dynamics of CH_4 emission from the reed marsh.

Before June 10th, the soil had a low moisture concentration because of the dry weather, and the reed field acted as a sink for CH_4, with the flux ranging between -968 and -29 μg/m^2/h. After the soil became saturated, in combination with the increase of temperature, the reed growth accelerated, and CH_4 emission remained positive until the end of October. Fig. 2 indicates that the reed marsh acts as a strong source of methane emission in summer and a weak source in autumn. Several temporary extremes appeared during this period. The average emission rate was 128–2,734 μg/m^2/h. Later, with the increase of dead roots, underground stems, and fallen leaves, the soil became less saturated, and the temperature also decreased. All these inhibited the activity of CH_4 bacteria, resulting in the decrease of CH_4 emission.

The seasonal dynamics of CH_4 emission is closely related to the rhizosphere of reed, because the underground stems criss-cross in the soil, which improves the soil air condition. In addition, the remnant underground roots can be greater than 10 t/ha (Hu, 1996). Their gradual decomposition creates a thick humus layer (with organic matters 3–6%) (Liu, 1984), which is the source of CH_4 emission.

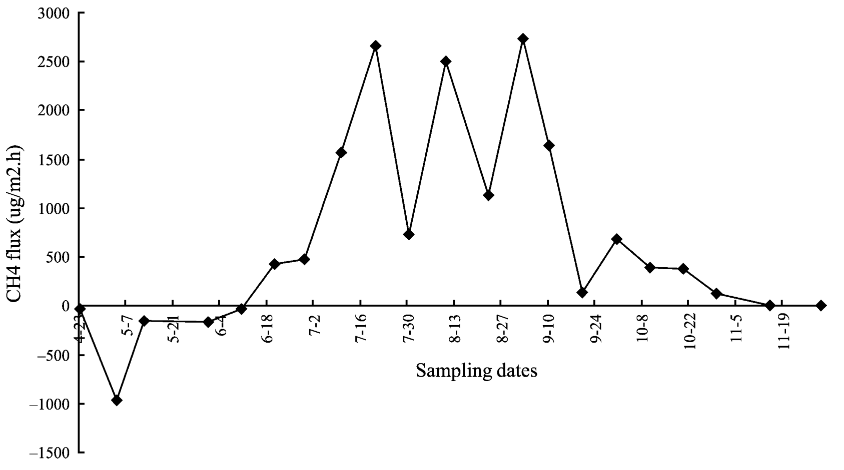

Figure 2: Seasonal dynamics of CH_4 emission.

The decomposition process is affected by many factors, such as temperature, water and nutrient condition, the freshness of remnants, as well as the activity of bacteria. Apart from the influence of the rhizosphere, irrigation can also be an important factor for CH_4 emission. The drainage water from paddy fields has a rich content of CH_4-inducing materials. All the above processes/factors affect CH_4 emission differently, each with a seasonal character.

3.5.2. The Effect of Reed Plants on CH_4 Emission

CH_4 produced in soil is emitted into the air mainly through the reed plant, and a minor part through air bubbles and molecular diffusion. Reed is a perennial gramineous plant, with many ventilating tissues in the leaves, sheathes, stems, underground stems and roots. When the reed is cut off above the water surface, CH_4 emission is only slightly reduced. On the other hand, when the reed is cut above soil surface, which is underwater, the CH_4 emission rate decreases by almost 60%. This means that about 60% of the methane produced in soil is transmitted into the air via reed plant. Otherwise it could be oxidized when passing through the water layer.

Earlier study reported that 90% of the CH_4 could be oxidized at aerated soil surface, while 11–100% of bio-originated methane could be oxidized in peat land (Sundh et al., 1995). Reed increases CH_4 emission not only by secreting organic matter into the rhizosphere via roots, but also by transporting CH_4 into the air via ventilating tissues, which largely reduces the oxidization of CH_4.

Furthermore on account of the stimulating effect of reed plants on CH_4 emission, inundated wetlands with reed growth emit much more methane (1,728 $\mu g/m^2/h$) than those without (115 $\mu g/m^2/h$). The difference can be 15 times.

According to our field measurement data, CH_4 emission in the natural wetland of Liaohe Delta was positively related to the seasonal dynamics of temperature, and negatively related to Eh value and water depth (Huang et al., 2001). The activity of methanogenic bacteria was higher in the rhizosphere and surface layer, and thus contributed more to CH_4 emission than other layers. It can be concluded from the above facts that reed plants play an important role in transportation, emission and production of methane in wetland soil.

3.6. Biodiversity Protection

The Liaohe Delta is situated on the migration route of some East Asia Avifauna. The wetland ecosystem provides abundant food resources and shelter sites, and thus becomes an important habitat for many residential wild animals, as well as a stopping place for migrating birds. The food resources such as fish, shrimp, crab, clam and seeds are widely distributed from coastal seawater and breeding ponds to inland reed and paddy fields. Habitat types include various reed communities, *Suaeda heteroptera* communities, *Aeluropus litoralis* communities and *Nitraria sibirica* communities. Now a National Nature Reserve of 80,000 ha has been established to protect this unique habitat for the wild species (Table 6).

Table 6: The biodiversity status of Liaohe Delta (Liu, 2000).

Item	Liaohe delta
Flora	Vascular plants: 224 species. Among them, pteridophyte: 1 family, 2 species; angiosperm: 33 families, 174 species.
Fauna	Terrestrial vertebrates: 63 families, 273 species; mammal: 11 families, 21 species; avifauna: 46 families, 238 species; reptiles: 3 families, 19 species; amphibious animals: 3 families, 4 species.
Avifauna	National first-grade protected avifauna: 4 species; national second-grade protected avifauna: 27 species; resident (birds): 40 species; summer resident: 77 species; winter resident: 11 species; migrating bird: 116 species. The northern most site for the breeding of saunder's gull (*Larus saundersi*), and southern most site for the breeding of sacred crane (*Grus japonensis*).
Fish	Freshwater fish: 16 families, 67 species. Among them, Syprinid: 36 species, taking 53.7%. Sea fish: 120 species; among them, bony fish: more than 100 species, taking 90%.

3.7. Conclusion

The wetlands in the Liaohe Delta play an important role in water regulation, nutrient and pollutant removal, bio-production, bio-conservation and many other aspects. Intensive human activities will largely affect the ecological functions of the wetlands. How to combine local economic development and nature conservation together will be a critical question to be answered in the future research projects.

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Chapter 4

The Dyke-Pond Systems in South China: Past, Present and Future

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Abstract. In 1990 inland aquaculture accounted for about two-thirds of total aquaculture production in the world, of which about three-fourths came from pond culture in China. This chapter is an attempt to review the past, present and future of the dyke-pond system widely adopted in the southern part of China. There has been a long-term tradition of using organic wastes in inland aquaculture in most of the Asian countries, and some East European Countries such as Hungry. Manure served as pond fertilizer to enrich nutrients in the pond water, which facilitate algal growth. Fungal growth on manure particles is also enhanced. These organisms will in turn serve as food for different fish species with different feeding modes, and, therefore, all the substances derived from the manure could be fully utilized. However, the recent rapid socio-economic changes in the region have resulted in the discharge of a large volume of domestic and industrial effluent which is untreated. Single species of high priced fish (monoculture) is cultivated using high protein grains instead of polyculture using manure as the major energy input.

In addition, chemical fertilizer has been replaced by the use of animal manure. Other chemicals used in aquaculture operations including sediment and water treatment compounds, pesticides, disinfectants, antibiotics, vaccines, immunostimulants, vitamins, etc. all exert harmful effects on the cultured fish, occupational health, adjacent ecosystems, food safety and human health. It is commonly known that persistent organic pollutants (POPs such as total polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), dichlorodiphenyltrichloroethane (DDTs)) and heavy metals and metalloids (such as lead (Pb), mercury (Hg), and chromium (Cr), arsenic (As)) can enter fish ponds and exert harmful effects. Antibiotics (such as streptomycin, chlortetracycline, oxytetracycline, tetracycline, etc.) included in feed additives are anticipated to end up in

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the environment. These chemicals may be taken up by fish, washed off into surface waters or leached to groundwater where they can adversely affect both environmental and human health. Investigations are needed to study the persistent behavior, distribution and ecotoxicological effects of these contaminants in the pond environment and adjacent aquatic ecosystems; the health risk due to the consumption of contaminated fish; and the possible evolution of antibiotic-resistant bacteria due to the overuse of antibiotics. This information will be useful for preparing the guidelines for “Good Aquaculture Practices” or “Organic Fish Farming” for aquacultural industries in our region.

4.1. Introduction

Aquaculture production was 21.8 million tonnes in 1998, which accounted for more than 55% of the total fishery production in China (Qi, 2002). Freshwater fisheries or inland aquaculture accounted for about two-thirds of the total aquaculture production, of which about three-fourths came from pond culture (Wang & Yi, 1995). These facts suggest that the future of aquaculture production in China will depend heavily on the pond aquaculture.

Being an agriculture based country, China has a long tradition of waste recycling and utilization. Recycling of wastes can utilize residual energy contained in the wastes, ease waste disposal pressure, and hence mitigate land and water pollution problems. Animal and human manure are mixed with plant residues such as rice straw to produce compost, which serves as soil conditioner and fertilizer for improving both physical and chemical conditions of nutrient deficient soils. Verimcomposting involves using earthworms to degrade organic materials, and the verimcompost produced also serves as a soil conditioner/fertilizer, in addition to the production of high-grade animal protein (earthworms). Anaerobic digestion of waste materials with appropriate C/N ratio fulfills the dual purpose of methane (biogas) generation for fuel, and maintenance of sanitation in the rural areas. The digested slurry is added to the fishponds as pond fertilizer for polyculture (rearing different fish species simultaneously) of fish, resulting in increased fish yields (Polprasert, 1996).

The “Feedlot” method using cereal grains or high protein feed pellets, rearing a single fish species (monoculture) is commonly practiced in Europe, USA and Japan. By contrast, there has been a long tradition of applying waste materials to freshwater fish ponds in China, most of the Asian countries and some of the East European countries such as Hungry. The use of wastes is due to the unavailability of grain feeds. The waste materials include animal and human excreta, kitchen wastes and other agricultural and industrial byproducts. Some of the waste materials such as kitchen wastes can be directly consumed by fish, whereas animal and human excreta serve as fertilizer to enrich nutrients in the pond water to facilitate growth of microorganisms such as fungi and algae, which in turn serve as food for higher trophic organisms, including some species of fish thriving in the pond. Polyculture involving different fish species, with carps as the major component, and waste materials as the major source of input has been practiced for almost four thousand years in China (Heilongjiang Aquacultural College, 1990). All the substances derived from the waste materials can be fully utilized by different aquatic organisms and fish species, with different feeding habits and digestive systems. This has led to very high fish yields.

However, due to the rapid socio-economic development during the past 20 years in the southern part of China, chemical fertilizer and high protein feed lots are used instead of waste materials, and very often monoculture of high priced fish is practiced, instead of polyculture. This coupled with the deterioration of water quality due to the contamination by nutrients and pesticides, casts doubt on the fish quality and the health of consumers. In fact our studies indicate that the human breast milk collected from Hong Kong and Guangzhou (the biggest city in South China) contained high concentrations of DDT and PCBs, which are related to their diets, with a high consumption of fish (Wong et al., 2002a,b). However, little is known about the relationship between the operation of pond aquaculture and environmental pollution in our inland aquatic ecosystem.

The objectives of this chapter include the review of (1) the dyke-pond system in South China; (2) the traditional practice of using waste materials for aquaculture; (3) the recent socio-economic changes in South China related to the freshwater aqucultural industries; and (4) recommendations for tackling the problems.

4.2. The Dyke-Pond System in South China

In the southern part of China including Hong Kong, a large part of swamps around the Pearl River Delta were gradually reclaimed for productive use some six centuries ago. Wetlands have been transformed to ponds separated by cultivable ridges (dykes) for growing different crops. Components of the system include mulberry, silkworm, vegetable and sugar cane, etc. Ponds are managed in a rotation of harvesting and stocking, with 4/5 fish species, and carps served as the major components. The organically enriched mud at the bottom of the pond is dredged regularly to fertilize the dyke soil for crop production. Due to the favorable climate and water resources, the region has been known as “homeland for rice and fish”. Table 1 compares the fish production of selected Asian countries.

Chinese fish farms produced two third of the world’s yield of farmed fish in 1990 (FAO, 1992). The average rates of sustained production throughout most of the world were in the range of 300-500 kg/ha, while it has been demonstrated in both

Table 1: Inland and coastal aquaculture production in some selected Asian countries/regions in 1990.

Country/region	Total production (t)	Inland production (t)	Coastal production (t)
Bangladesh	169,758	151,161	18,624
China	7,200,383	4,204,728	2,995,655
Hong Kong	10,256	5,525	4,731
India	1,011,136	982,136	29,000
Indonesia	558,795	242,625	316,170
Korea, DPR	208,670	11,200	196,470
Korea, Rep. of	789,765	15,823	773,942
Malaysia	47,876	7,007	40,869
Pakistan	40,057	40,016	41
Philippines	672,316	81,127	591,189
Sri Lanka	5,700	5,000	700
Taiwan	343,954	148,795	195,159
Thailand	253,326	92,466	160,860
Vietnam	155,000	123,000	32,000

Source: FAO, 1992.

China and Israel that, it is possible to achieve experimental production rates as high as 18,000 kg/ha/year (Advisory Committee on Technology Innovation, 1981).

4.2.1. Integrated Agricultural and Aquacultural Systems

Common waste materials include municipal wastes, sewage sludge, agricultural wastes, coal ash, toxic, hazardous and difficult wastes, but not all of them are suitable for recycling in aquaculture. Waste materials that could be used as fish feeds include both agricultural (such as rice bran) and industrial (such as food processing wastes and brewery wastes). Manure can be regarded as pond fertilizer because nutrients such as nitrogen and phosphorus will be released upon decomposition to facilitate algal growth. In addition, fungi can also make use of manure as their growth substrate. These microorganisms will in turn serve as food for different fish species. Some species such as tilapia may consume manure directly.

In some areas, the effluent from waste stabilization ponds is introduced to fishponds where herbivorous fish species are cultivated, and sometimes these fish are also cultivated in the stabilization ponds themselves, in order to remove algae and to upgrade the effluent (Schroeder, 1975). Our early study (jointly with the Agriculture, Fisheries and Conservation Department) rearing a mixture of freshwater fish (silver carp, big head, common carp, grass carp, tilapia and black bass) in four fishponds receiving polluted river after it has been treated by sedimentation and aeration, indicated that all fish (except grass carp) grew to marketable size within one year (Liang et al., 1999a).

The integrated agricultural and aquacultural system with animals such as pigs and ducks reared in the vicinity of the fishponds enables the easy delivery of manure, avoids nutrient loss during transportation, and cuts down transportation cost. Ducks release manure directly to the ponds, and also keep the water surface clean by consuming algal blooms and higher plants such as duckweeds. Combined fish and duck culture is widely practiced in Eastern Europe, especially Hungary, with improved feed conversion, increased fish yield, as well as improvements in both duck quality and yield (Advisory Committee on Technology Innovation, 1981).

4.2.2. General Principles of Using Manure in Polyculture of Fish

The common fish species used in polyculture include the following, each with distinctive feeding mechanism: (1) Grass carp *Ctenopharyngodon idellus* eats all kinds of food, although higher plants, submerged grasses and detritus are its major food items; (2) Silver carp *Hypophthalmichthys molitrix* is also a filter feeder, mainly consuming phytoplankton; (3) Big head, *Aristichthys nobilis* is able to trap zooplankton whilst swimming with its mouth wide open; (4) Common carp *Cyprinus carpio* is omnivorous and eats plants and snails; (5) Black carp *Mylopharyngodon piceus* eats snails, aquatic insects and crustacean; and (6) Mud carp *Cirrhinus molitorella* feeds on benthic organisms such as worms, shrimps and detritus. Cultivation of different carp species is of a great interest not only in terms of available food utilization but also with respect to the utilization of all the ecological niches available in the pond system, as surface feeder, column feeder or bottom feeder (Table 2). Sometimes tilapia *Tilapia mossambicus* (also omnivorous and eats detritus, plants and plankton) is added as the association of carp, and tilapia may increase the growth of carp (Hepher, 1988).

The use of a polyculture of mixed fish species, which is adapted to feed on various organisms can fully utilize all the substances derived from the wastes. The synergistic interactions among fish species reared in polyculture are clearly explained by Milstein et al. (1995): faecal pellets, rich in partially digested phytoplankton, are discharged from silver carp than consumed by the common carp which otherwise could not use these algae. The common carp in turn recirculates nutrients into the water column by stirring up the mud, and interferes

Table 2: Feeding habits and ecological niche of the different carp species.

Species	**Feeding habits**	**Ecological niches**
Grass carp, *Ctenopharyngodon idella*	Macrophytes	Surface
Silver carp, *Hypophthalmichthys molitrix*	Phytoplankton	Surface
Big head, *Aristichthys nobilis*	Zooplankton	Mid-water
Common carp, *Cyprinus carpio*	Benthos + detritus	Bottom
Black carp, *Mylopharyngodon piceus*	Benthos	Bottom
Mud carp, *Cirrhina molitorella*	Benthos + detritus	Bottom

with the development of filamentous algae and higher plants, thereby raising phytoplankton production and hence food for silver carp. Understanding of the dynamics of the natural food web is deficient, because most studies have been related to pond fish culture based on the gut contents of fish, or measurements of primary production and the standing stock of plankton.

4.2.3. Nutrient Dynamics of Fish Ponds Using Manure as the Major Input

The high yield of fish using this system is mainly due to rational use of manure, good management practices, suitable climate and water supply. Fish grow rapidly in warm weather, and wastes can replace the need for feeds. This gives rise to higher yields of high-grade animal protein at a lower cost, with a better (lower) food conversion ratio in comparison with other farmed livestock (Table 3). In addition, it is envisaged that the accumulation of metabolic waste in polyculture fishponds is less than monoculture ponds.

Table 3: Efficiency of feed utilization of various animal species per 1,000 g of feed intake.

Species	**Live weight gain (g)**	**Food conversion ratio**	**Energy gain (kcal)**	**Protein gain (g)**
Chicks	356	2.8	782	101
Pigs	292	3.4	1,492	30
Sheep	185	5.4	832	22
Channel catfish	715	1.4	935	118
Brown trout	576	1.7	608	75

Source: Hastings & Dickie (1972).

Direct consumption of manure usually results in lower yield due to the lower energy and protein contents contained in the manure. The advantage of applying manure to fishponds is mainly its role as a pond fertilizer for autotrophic production via photosynthetic production of plankton, or as a base for the heterotrophic production of bacteria and protozoa which are utilized by pelagic and bottom-feeding fish. There seems to be insufficient information relating to the flow of manure and its metabolic products through the fishpond ecosystem, therefore, more information on various pathways of the natural food web would aid the understanding of nutrient dynamics in the system. Nevertheless, it is understood that the advantages of using manure in fish culture include: (1) stocking and harvesting different fish species in a rational manner; (2) decreased hazard of fish killed due to anoxia; (3) more efficient use of nutrients; and (4) stable water quality due to higher oxygen concentration and higher pH; and all of these factors contribute to higher fish yields. In other words, ecological balance is achieved between production and consumption. Algae, bacteria, and fish form symbiotic relationships in a well function waste-fed fishpond (Bhattarai, 1985).

It is commonly known that the use of raw or untreated organic waste imposes adverse effects on treated organisms, mainly due to the release of ammonia, and other volatile substances. Pretreatment of wastes is, therefore, essential. Anaerobic digestion involving the mixture of pig manure and rice straw can (1) generate biogas for fuel, (2) sanitize the manure as the high temperature of the digestion process destroys eggs of parasites, and (3) treat the manure so as to avoid harmful substances such as ammonia being released into the pond water. Aerobic decomposition involves a group of microorganisms including bacteria, fungi and actinomycetes, and by providing suitable temperature, moisture and aeration, the digested materials should be free of harmful substances, and can be recycled in the pond system.

4.3. Recent Socio-Economic Changes and Their Effects on the Aquacultural Industries

4.3.1. South China and Persistent Organic Pollutants (POPs)

Being the first "economic zone" in the whole of China, the region of Hong Kong and South China has undergone rapid socio-economic change during the past 20 years. Population growth, urbanization and industrialization resulted in the deterioration of water quality in most streams and rivers, due to the discharge of untreated domestic and industrial effluent. The change of land use is also an important problem as good agricultural land has been used for the constructions of highways and housing estates. Marginal land, which has been reclaimed for agricultural purpose, has required a large amount of chemical fertilizer, so that the runoff results in eutrophication along coastal areas (Neller & Lam, 1998). The traditional practice of organic waste recycling in both agriculture and aquaculture is diminishing. These include the declining use of waste materials for biogas generation. The accumulation of waste materials, especially animal manure, therefore, results in further land and water pollution problem.

Since the adoption of the Stockholm Convention on POPs, worldwide attention by scientists, policy-makers, industries, NGOs, and the general public on environmental health and management issues relating to POPs has been increasing. The 12 Stockholm POPs include pesticides (such as aldrin, endrin, DDT, etc.), industrial chemicals (such as PCBs) and unintentional by-products (such as dioxins/furans). Most of these chemicals are toxic, long-lived, can travel long distances, and move from warm areas to colder areas. Due to their affinity to lipids, they are absorbed by the fatty tissues of animals and humans and are bioaccumulated and biomagnified through the food chain. There is a severe lack of information related to the sources, fates and effects of POPs in our region (Wong & Poon, 2003).

Heavy metals, fuels and lubricants originating from industrial activities, dumps or human settlements near fishing farms, as well as pollutants from non-point sources such as pesticides from agricultural run-off can be present in the ponds, usually due to non-intentional contamination (Boyd & Massaut, 1999). Freshwater sediments collected from the inland river systems and fish ponds in the Pearl River Delta were grossly polluted by various heavy metals, PCBs, HCHs and DDTs, which resulted in higher concentrations of these chemicals in fish collected from inland rivers as well as from fish ponds, with fish collected from inland rivers having higher concentrations of these contaminants (Zhou et al., 1998, 1999a; Zhou & Wong, 2000). It is also alarming to discover that the PCB concentrations in muscle tissue of grey mullet (*Mugil cephalus*) collected from a nature reserve (Mai Po Marshes), a remote area, exceeded the guideline of 0.01 μg/g wet weight basis, imposed by US EPA for human consumption (Liang et al., 1999b).

The uptake of contaminants by fish seemed to depend on their feeding modes, with carnivores having higher concentrations. Table 4 indicates that black bass, which are located in the highest trophic level, contained the highest levels of DDTs, PCBs and Hg (Zhou et al., 1999b; Zhou & Wong, 2000). The quality of fish and their safety for human consumption has recently become a controversial issue. Our recent study revealed that the higher concentrations of organochlorine pesticides (DDT, DDE and HCH) and PCBs contained in human breast milk collected from two populations (Hong Kong and Guangzhou, the largest city in South China) were related to diet, especially fish consumption (Wong et al., 2002a,b). This may well be harmful to the next generation, as infants borne to

Table 4: Contaminant concentrations in fish species related to their feeding modes.

Species	DDTs (µg/g, lipid)	PCBs (µg/g, lipid)	Hg (ng/g, dw)	Feeding habits
Black bass, *Micropterus salmoides*	0.760	3.4	56.7	Carnivorous, shrimps and mosquito fish
Tilapia, *Oreochromis mossambicus*	0.090	3.1	13.7	Omnivorous, algae, detritus, benthic invertebrates, small shrimps
Common carp, *Cyprinus carpio*	–	–	18.9	Omnivorous, benthic invertebrates, aquatic insect larvae
Big head, *Aristichthys nobilis*	0.040	0.87	33.8	Filter feeder, zooplankton
Silver carp, *Hypophthalmichthys molitrix*	0.038	1.6	20.8	Filter feeder, phytoplankton
Grass carp, *Ctenopharyngodon idellus*	0.087	2.1	26.3	Herbivorous, macrophytes
Shrimps	–	–	13.3	Algae, zooplankton

– indicates not tested; Zhou et al., 1999b; Zhou & Wong, 2000.

mothers with high body concentrations of dioxin-like PCBs can experience low birth weight (Brouwer et al., 1998). In our region, bioaccumulation and biomagnification of persistent toxic substances within aquatic ecosystems will increase the risks to inhabitants with a strong preference for consuming freshwater fish.

4.3.2. Environmental Impacts of Inland Aquaculture

Basically the different environmental impacts caused by inland aquaculture can be described in terms of the amount of excreted metabolites and uneaten food, and of the chemicals used during fish production (Papoutsoglou, 1992). However,

Table 5: Adverse environmental impacts of aquaculture on the environment.

Physicochemical and biological changes	Reference
Modification of water temperature and flow rate profiles	Billard & Perchec (1993)
Increased concentration of suspended solids, BOD, COD, forms of N (including NH_3), P	Warrer-Hansen (1982)
Reduced concentration of DO	Bergheim & Silvertsen (1981)
Alteration of drinking water by use of chemicals and antibiotics	Buchanan (1990)
Generation of organic-rich sediments	Holmer (1992)
Occurrence of algal blooms in eutrophic waters	Gowen et al. (1990)
Modification of the biotic index (based on invertebrate communities)	Gowen et al. (1988)
Genetic pollution	Hepher & Pruginin (1981)
Increased risk of disease spread	Hubbert (1983)
Conflicts with other usages: fishing, agriculture and recreational activities	Gowen (1992)

the impacts on the surrounding environment are more diversified, and include causing physicochemical and biological changes to the surrounding areas, as well as conflicts with other economic and recreational activities (Table 5).

A number of chemicals such as fertilizers, liming material, disinfectants, antibiotics, algaecides, and herbicides are used in pond aquaculture for improving soil and water quality, and for controlling biological problems such as phytoplankton blooms, aquatic plant infestations, disease vectors, and the proliferation of wild fish (Boyd & Massaut, 1999). The most common chemicals used in inland aquaculture fall into three classes, according to Alderman & Michel (1992):

(1) Topical disinfectants including a wide and diverse range of compounds, such as malachite green, formalin, salt, copper sulphate, potassium permanganate and quaternary ammonium compounds, used in the treatment of topical parasites (including some bacteria, protozoa and fungi).
(2) Organo-phosphates, primarily dichlorvos, for eliminating crustacean predators of fish fry in cyprinid nursery ponds.
(3) Antimicrobials which include a wide and diverse range of compounds listed in Table 6.

Pesticides are intentionally applied to ponds in order to kill unwanted organisms before stocking with fish or shrimp. For example, organophosphate pesticides that

Table 6: Major antimicrobial drugs used in aquaculture.

Drug		Route	Dose/application interval	Indication
Antibiotics	*Product*			
β Lactams	Ampicillin	Oral	50–80 mg/kg 10 days	Gram-negative bacteria
	Amoxycillin	Oral		
Aminoglycosides	Neomycin	Oral	50–80 mg/kg 10 days	Gram-negative bacteria
	Kanamycin	Bath	20 mg/kg	
Tetracyclines	Tetracycline	Oral	50–80 mg/kg 10 days	Gram-negative bacteria
	Oxytetracycline	Bath	20 mg/kg	
	Doxycyline			
Macrolides	Erythromycin	Oral	50 mg/kg 10 days (bathe eggs in 2 mg/kg 1 h)	Bacteria kidney
Non-classifiable	Chloramphenicol	Oral	50–80 mg/kg 10 days	Gram-negative bacteria
		Bath	20 mg/kg	
Synthetic antibacterial	*Agent*			
Sulphonamides	Sulphamethazine	Oral	200 mg/kg 10 days	Gram-negative bacteria
	Sulphadimethoxine			
	Sulphaguanidine			
Potentiated sulphonamides	Trimethoprim + sulphadiazine	Oral	50 mg/kg 10 days	Gram-negative bacteria
Nitrofurans	Furazolidone	Oral	50–80 mg/kg 10 days	Gram-negative bacteria
	Furaltadone			
	Nifurpirinol	Oral	10–50 mg/kg 10 days	
		Bath		
Quinolones	Oxolinic acid	Oral	12 mg/kg 10 days	Gram-negative bacteria
	Flumequine			

Alderman & Michel (1992).

are used against trematodes or mysids pose a major threat to exposed non-target crustaceans (GESAMP, 1997). In a Norwegian study, Egidius & Møster (1987) demonstrated that Neguvon (active ingredient: trichlorfos) treatment in salmon farms was the likely agent that caused the death of lobsters and the disappearance of crab populations adjacent to salmon farms. The heavy metals which present the greatest threat to contamination of aquaculture products are lead, mercury, arsenic, beryllium, cadmium, chromium, manganese, silver, and zinc (Boyd & Massaut, 1999): these substances can be toxic to the cultured species, and they can contaminate the harvested product thus imposing potentially adverse effects on human health.

Due to the socio-economic changes, high priced fish are in greater demand, leading to monoculture and the use of high protein feed grains. Antibiotics are very often used to increase the immunity of fish in order to help them survive in poorer water quality. As in other countries, the main groups of antibiotics used in pond aquaculture are tetracyclines, sulfonamides and chloramphenicol (Migliore et al., 1993). These are applied through the feed or by simple addition to the water. Most of the antibiotics in excess feed end up in the sediments where they are either degraded (Lai et al., 1995) or slowly leached back into the surrounding water (Smith & Samuelsen, 1996). It has been estimated that about 70% of the antibacterial agents applied in fish farming are released into the environment (Schneider, 1994).

In 1991, the Japanese Health Authority found unacceptable levels of oxytetracycline in farm raised shrimps imported from Thailand (Weidner & Rosenberry, 1992). In a study conducted in 1990-91, 8% of the shrimps *Penaeus monodon* (over 1400 samples) obtained from Bangkok markets contained residues of tetracyclines, quinolones, sulphonamides and penicillins (Saitanu et al., 1994). In the USA, analyses for chloramphenicol in imported shrimps have been conducted regularly. From 1992 to 1993, it was found that five samples (three were obtained from Thailand and two from China, 3.2% of the samples tested) contained measurable amounts of chloramphenicol (Weston, 1996).

Tetracyclines, oxolinic acid, several sulpha drugs and trimethoprim are all known to be persistent, with varying results in degradation studies depending on temperature, depth in sediment, etc. (Halling-Sørensen et al., 1998; Weston, 2000). Several antibiotics, e.g. oxytetracycline, oxolinic acid and flumequine, have been found in sediments 6 months after treatment (Weston, 1996; GESAMP, 1997). The half-life of oxytetracycline was estimated to be 10 weeks in anoxic sediment (at 4–8°C) in a model of a fish farm bottom (Jacobsen & Berglind, 1988). There is a lack of local information concerning the presence of antibiotics in fish ponds and adjacent aquatic ecosystems.

It is widely recognized that the extensive use of antibiotics in agricultural animal production contributes to the development of antibiotic-resistant pathogens, and that these microbes can infect both humans and domesticated animals (Willis, 2000). Development of resistant pathogens in aquaculture environments was documented (Sørum, 1999), and evidence for transfer of resistance encoding plasmids between aquaculture environments and humans was presented (Rhodes et al., 2000). Accumulation of antibiotics in sediments may interfere with bacterial communities and affect the mineralization of organic wastes (Stewart, 1994). Digested pig manure has been traditionally used as pond fertilizer for culturing different carp species (polyculture of common carp, grass carp, big head, etc.) in the region, and the effect of antibiotics associated with animal manure on pond culture is unknown. In addition, a large amount of antibiotics has been used for monoculture of some species (such as mandarin fish, a carnivore) with a higher economic value

Some recent studies also indicate that several of the antibiotics, e.g. ciprofloxacin, oxolinic acid, chlortetracycline, oxytetracycline, tetracycline, tiamulin and trimethoprim, are acutely toxic to algae and aquatic invertebrates (Halling-Sørensen et al., 1998; Wollenberger et al., 2000). The toxic effect data of antibacterial agents on various aquatic species reported in the literature (Macri et al., 1988; Migliore et al., 1993), show values in the mg/l range. To make an ecotoxicological risk assessment, the concentrations of potentially hazardous chemicals in the environment should be compared with concentrations of chemicals for which biological effects have been reported.

4.4. Good Aquacultural Practices and Organic Fish Farming

A recent survey on the state of world fisheries and aquaculture by FAO (2002) recognized that China is the world's largest producer of farm-grown aquatic products today, and that the main challenges to further development are the limited supply of good quality seeds for some fish species; the oversupply of traditionally cultured species; the under exploitation of high value species; outdated farming technologies; water pollution; the limited availability of suitable land for expansion; and frequent fish disease outbreaks. To overcome these, it was recommended to (1) develop industrialized farming systems by improving the design and upgrading production, employ the latest technology and select the best combination of species to respond to market conditions in China and abroad; (2) raise the market share of high-value freshwater species suitable for export, and achieve production efficiency through the adoption of large scale industrial farms; (3) pay emphasis to the production of high quality seed by making use of modern biotechnology; and (4) establish an integrated scientific system and network of fish breeding and seed production for high-quality indigenous or endemic species, as well as fish health management to improve disease prevention, diagnosis, control and treatment.

All the aforementioned factors should be explored using the existing models available in China, taking into consideration agricultural and aquacultural development, land use and coastal management, pollution control, and food quality and safety. Aquaculture is an ancient practice and until recently has developed mainly on a trial and error basis. However, it is commonly recognized that the addition of aquaculture into the mix of farm enterprises can greatly raise the efficiencies of bioresource flows and profitability (Ruddle & Zhong, 1988), and there are different successful models of ecologically integrated freshwater systems available in different parts of the world (Table 7).

Although guidelines are available for the use of different chemicals in aquaculture, and for the residual levels of different antibiotics and synthetic antibacterial agents in aquacultural products in China (Standards on Agricultural Industries of P. R. China, 2002), very often they are not fully enforced. Regulation is also needed to control the impact of aquaculture development on the environment. The approaches to the regulation of aquaculture for environmental protection should include: (1) development permits, specifying size and/or location of the fish farm; (2) environmental quality standards which could be achieved through nutrient loading limits and other conditions related to discharge; (3) waste minimization regulations, related to feeding management; and (4) discretionary requirement for environmental impact assessment. In order to reduce operational costs, and promote "green" marketing opportunities, it is essential to adopt environmental management systems consisting of environmental review, environmental policy, system design and implementation, and environmental audit in order to offer a strategic framework for the control of environmental impact from inland aquaculture, and for improving environmental performance in the aquaculture business (Milden & Redding, 1998).

Like "organic farming", the demand for "organic fish farming" is increasing worldwide, e.g. production of organic salmon in Scotland. Some of the regulations stipulated for organic salmon farming (Costa-Pierce, 1988) included: (1) density of fish should be less than 10 kg/m^3 (compared with traditional salmon farming of 20 kg/m^3); (2) fish feed has to come from certified feed manufacturers, and must contain no genetically modified ingredients, no artificial colors, no fish meal from industrial fishing (fish meal produced from fish processing wastes is used), and all cereals used have to come from organic farming; (3) fish should be fed by hand; (4) no carbon dioxide is allowed in slaughtering; (5) no anti-fouling chemicals are allowed in treating nets; and (6) no polystyrene boxes are allowed for marketing the fish (waxed cardboard or reusable plastic boxes are used). Most important of all is the close monitoring on the use of chemicals, and on the quality of the

Table 7: Some examples of ecologically integrated freshwater systems.

Systems descriptions	**Reference**
Chinese dike-pond aquaculture ecosystem integrates aquaculture, plant and animal agriculture, silviculture and sericulture, sustainably producing 20–40 t/ha/yr	Ruddle & Zhong (1988) and Korn (1996)
Silvofisheries: integrated mangrove forest aquaculture systems, e.g. dyke-pond system in South China including Hong Kong	FitzGerald (1988)
Polyculture of carps, mullet and prawns, with effluents irrigating a mixed tropical orchard	Costa-Pierce (1987)
Mixed intensive tilapia culture in tanks integrated with hydroponics producing commercially viable fish and plant yields and using sludges for pastures	Rackocy & Hargreaves (1993)
Reuse of saline aquaculture effluents to irrigate halophytes suitable for forages	Brown & Glenn (1999)
Effluents from channel catfish ponds coupled with bulrush, cutgrass and maidencane aquatic wetlands achieved excellent removal of all nutrients, BODs and solids	Schwartz & Boyd (1995)
Biculture cage ecosystems (one species being grown in a culture system receiving formulated feeds placed above or at higher elevation from a second system below it holding species that are unfed)	Costa-Pierce & Hadikusumah (1990)
Tilapia aquaculture integrated into irrigation schemes in the US southwest	Olsen & Fitzsimmons (1994)

aquacultural products which are relatively free of contaminants, and thus safeguard environmental and human health.

4.5. Conclusion

In view of the recent socio-economic changes in South China, it is essential to utilize organic wastes as much as possible. Integrated farming systems involving agriculture as well as aquaculture should be encouraged. Organic wastes should be applied to land in order to improve both the chemical and physical properties of poor soils, and as pond fertilizer in fish polyculture. Nevertheless, waste and wastewater treatment facilities should be constructed in South China, together with close monitoring of pond water quality. Laws relating to the use of chemical fertilizer, pesticides and other undesirable chemicals, such as antibiotics and growth hormones, should be established and then enforced within in the region through joint agreements between the Guangdong and Hong Kong governments. The availability of more data and information on the chemicals used in fish farming would be valuable for effective environmental and public health management. It is likely that the use of antibiotics and hazardous chemicals could be significantly reduced, without decreasing production yields, by disseminating correct information among farmers about the safe and effective use of antibiotics and other chemicals in fish farming. “Good Aquaculture Practices” and “Organic Fish Farming” should be established, in order to safeguard the health of inhabitants in the region, as well as to ensure that export products meet the standards imposed by other countries.

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Session II

Wetland Biogeochemistry

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Chapter 5

Heavy Metal Mobility and Aquatic Biogeochemical Processes at Mai Po Marshes Nature Reserve, Hong Kong

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Abstract. Mai Po Marshes is a sub-tropical estuarine wetland in Hong Kong, which is a complex, dynamic and unique ecosystem. The main objective of the present study is to explore heavy metal mobility in the sediments as a function of biogeochemical conditions in the aquatic environment in Mai Po Marshes. Samples of sediments and water were collected monthly from eight sites during July–September 1997. Heavy metal (Cd, Cr, Cu, Ni, Pb, Zn) concentrations in the sediments and their associations with aquatic physicochemical properties were examined. Aquatic biogeochemical processes appear to be different between landward and seaward sides. Heavy metal mobility seems to be closely associated with organic matter degradation on the landward side, while water quality and changes in redox conditions appear to be the major processes responsible for heavy metal mobility on the seaward side. At Mai Po Marshes, whereas organic enrichment benefits bird conservation as previous studies indicated, the results of this present study suggest that organic dynamics associated with heavy metal mobility and toxicity may also be taken into consideration for organic matter management.

5.1. Introduction

The Mai Po Marshes Nature Reserve, one among the 21 *Ramsar* sites in China, is located on the eastern shore of Deep Bay (part of the Pearl River estuary) in Hong Kong (Fig. 1). Its ecological importance has been extensively described (Hodgkiss, 1986; Irving & Morton, 1988). The reserve serves as an important winter ground for 24% of the world population of the endangered species, the black-faced spoonbill (*Platealea minor*) (Young, 1995). The major ecosystems at Mai Po Marshes include mangrove swamps, inter-tidal mudflats, shrimp shallow

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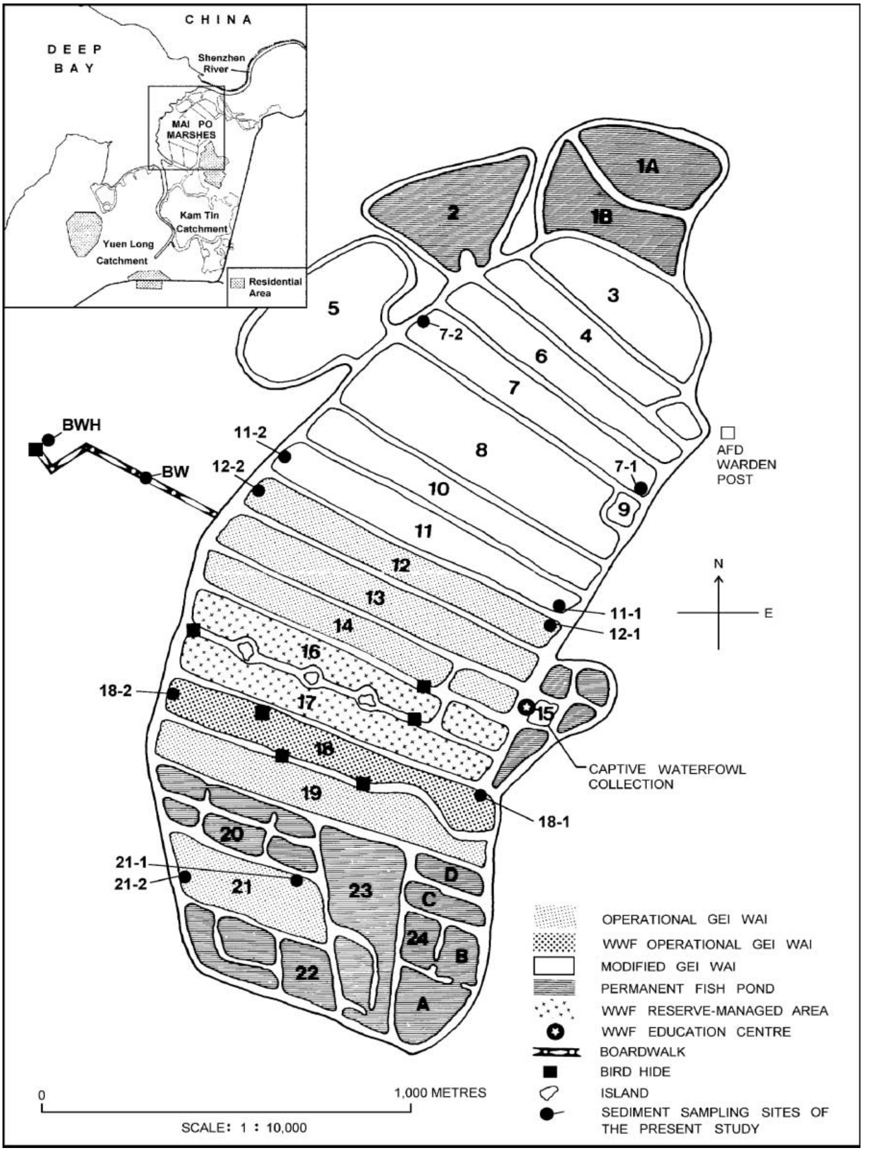


Figure 1: Map of the study sites showing the sampling locations.

ponds (local name *gei wais*) (Li & Lee, 1998) and freshwater fishponds. They are important nursery grounds for a number of migratory birds (Young, 1995).

Due to the discharges of domestic sewage, livestock wastewater and industrial effluent into Deep Bay from the Pearl River, Shenzhen River and Shan Pui River, Mai Po Marshes have been facing increasing pressure due to heavy metals pollution in the recent years. As to the inter-tidal mudflats, trace metals (Cd, Cr, Cu, Fe, Pb and Zn) were 4–25% more enriched in the upper 0–10 cm of the topsoil compared to that in the 21–30 cm core of the sediments, which indicates an increased anthropogenic input of metals in the sediments (Tam & Wong, 2000). In mangrove swamps, the surface sediments were seriously polluted by heavy metals (Cd, Cu, Ni, Pb and Zn) (Tam & Wong, 2000). As to *gei wais*, topsoil enrichment of Zn was revealed in the upper 0–8 cm compared with the bottom 8–16 cm, which indicates an increasing pollution in *gei wais* (Lau & Chu, 2000). Among the three ecosystems at Mai Po Marshes that have been studied, mudflats, which are the closest to Deep Bay, have the greatest levels of trace metals in the sediments, suggesting that the increasing heavy metal input most possibly comes from Deep Bay.

On the other hand, organic matter enrichment in Deep Bay may be essential for shorebird conservation. Anthropogenic input of organic matter is an important carbon source (up to 50%) for shorebirds (Li & Lee, 1998), and local catchments, such as Shenzhen River and Shan Pui River, provide the major nutrients for shorebirds at Deep Bay (Lee, 2000).

It is generally accepted that organic matter is one of the major controlling factor for heavy metal transportation in the sediment (Förstner, 1995; Schulin et al., 1995; Sauvé et al., 2000; Cantwell & Burgess, 2001; Chapman & Wang, 2001). Biological processes in the aquatic environment, such as organic degradation, nitrification and algal growth, play key roles in adjusting pH and Eh as well as in organic decomposition and formation (Wetzel, 2001). At Mai Po Marshes, it has been shown that the reduction of Mn (hydr)oxides was the major remobilization mechanism for metals such as Cd, Cr, Cu, Pb, Zn, Fe and Mn (Yu et al., 2000), but few studies have addressed the relationships between organic matter related processes and heavy metal mobility.

The main objective of the present study is to investigate heavy metal mobility and its associations with aquatic biogeochemical processes in water and sediments in *gei wais* at Mai Po Marshes. Six heavy metals (Cd, Cr, Cu, Ni, Pb and Zn) were selected because they have been the most studied anthropogenic pollutants (Ong Che & Cheung, 1998; Tam & Wong, 2000; Yu et al., 2000). Two locations in *gei wais* (landward and seaward sides) including four sites at each location were selected. Based on our field observation, sampling sites at the landward side had shallower and static overlying water (<0.5 m) above the sediments with more mangrove tree coverage, while those at the seaward side had deeper and flowing overlying water (1–1.5 m) with less mangrove trees. Through comparisons between the two locations, the major aquatic biogeochemical factors that influence heavy metal mobility will be identified. Information from the study may help further explore the mechanisms of heavy metal mobility and toxicity as well as organic matter management at Mai Po Marshes.

5.2. Materials and Methods

Detailed descriptions of the study site (Fig. 1) can be referred to in Liang et al. (1999). Monthly water and sediment samples were collected from the landward sites (7-1, 11-1, 12-1, 18-1) and the seaward sites (7-2, 11-2, 12-2, 18-2) during July–September 1997. Detailed methods of physicochemical analysis of water and sediments, as well as the total metal contents in the sediments, are described by Liang & Wong (2003). In general, besides water temperature (°C), pH, dissolved oxygen (DO) ($mg\,l^{-1}$) and salinity (%), four groups of parameters indicating biological processes in the overlying water were measured (APHA, 1985). These included: (1) organic matter metabolism related: biochemical oxygen demand ($mg\,l^{-1}$) (BOD) and dissolve organic carbon ($mg\,l^{-1}$) (DOC); (2) N metabolism related: NH_3–N ($mg\,l^{-1}$), NO_2^-–N ($mg\,l^{-1}$) and NO_3^-–N ($mg\,l^{-1}$); (3) P metabolism related: *ortho*-P (PO_4^{3-}–P) ($mg\ l^{-1}$) and total P (TP) ($mg\,l^{-1}$); and (4) biomass or particulate organic matter in water related: volatile suspended solids (VSS) ($g\,l^{-1}$) and total suspended solids (TSS) ($g\,l^{-1}$). Parameters representing major factors in the sediments influencing heavy metal mobility were measured, which included pH, redox potential (Eh) (mV), total organic matter (TOM) (%) and electrical conductivity (EC) ($mS\,cm^{-1}$).

Sequential extraction for heavy metal speciation analysis was performed according to the scheme proposed by Stover et al. (1976). The extracted heavy metal phases, reagent strength, extraction duration and solution to solid ratios are summarized in Table 1. Three replicates were carried out for each sediment sample. The filtrates of each phase were analyzed for heavy metals using Flame Atomic Absorption Spectrometry (Varian SpectrAA-20 model).

Statistical Analysis System for windows V8 was used for all statistical analyses. PROC ANOVA procedure was used for analysis of variance (ANOVA). PROC

Table 1: Summarized sequential extraction of heavy metals in the sediments.

Speciation phases	**Extractants**	**Reagent strength (M)**	**Extraction duration**	**Solution to solid ratio**
Exchangeable (Me1)	KNO_3	1	16 h, 200 rpm	50 ml:1 g
Adsorbed (Me2)	KF	0.5	16 h, 200 rpm	80 ml:1 g
Organic matter (Me3)	$Na_4P_2O_7$	0.1	16 h, 200 rpm	80 ml:1 g
Carbonates (Me4)	Na_2EDTA	0.1	8 h, 200 rpm	80 ml:1 g
Sulfides (Me5)	HNO_3	6	16 h, 200 rpm	50 ml:1 g

Stover et al. (1976).

CORR procedure was used for Pearson correlation analysis. PROC REG procedure was used for regression analysis.

5.3. Results and Discussion

5.3.1. Comparisons of Physicochemical Properties of the Water and Sediments Between the Landward and Seaward Sides in Gei Wais at Mai Po Marshes

Physicochemical properties of water and sediments in *gei wais* are shown in Table 2. One-way ANOVA results indicate that the landward side had significantly ($p <$ 0.05) higher organic matter (TOM) contents in the sediments (12.7%) than the seaward side (8.16%). Mangrove plants such as *Kandelia candel* and *Phragmites australis*are major producers in *gei wais*, contributing more than 95% of organic matter (t C yr^{-1}) production (Li & Lee, 1998). It is suggested that the uneven distribution of mangrove trees in *gei wais* caused the difference of organic matter in the sediments. Therefore, the higher TOM contents in the sediments of the landward side may be due to the more densely populated mangrove trees at the landward side. The significantly ($p < 0.05$) lower water temperature (29.4°C) at the landward side compared with that at the seaward side (30.5°C) might also possibly be the result of the more mangrove tree canopy at the landward side.

Due to the release of organic acids such as humic acid, pH levels can be reduced during the process of organic matter decomposition (Wetzel, 2001). The higher organic matter contents in the sediments at the landward side may be responsible for the significantly ($p < 0.05$) lower levels of pH (sediments: 6.99; water: 6.97) compared to those at the seawards side (sediments: 7.75; water: 7.48).

Compared with the landward side (TSS: 0.0229 g l^{-1}; PO_4^{3-} –P: 0.0972 mg l^{-1}), the seaward side had significant higher turbidity and *ortho*-P (TSS: 0.0469 g l^{-1}; PO_4^{3-} –P: 0.253 mg l^{-1}). Extremely high sedimentation (1.7 cm yr^{-1}) in *gei wais* (Lee, 1990) has been reported, and the tidal water from Deep Bay has been the major source of suspended solids and organic matter input to *gei wais* (Li & Lee, 1998). Besides the tidal water, potentially higher resuspension of sediments at the seaward side may possibly cause the higher TSS in the overlying water, since the seaward side is both the inlet and outlet of water flux in the *gei wais*.

5.3.2. Comparisons of Aquatic Biological Processes Between the Landward and Seaward Sides in Gei Wais at Mai Po Marshes

The correlations among aquatic physicochemical variables in *gei wais* are shown in Table 3. At the landward side, the correlation results suggest that organic matter

Table 2: Summary of physicochemical properties of the water and sediments in *gei wais* at Mai Po Marshes.

	Landward			Seaward		
	Mean ± SD	**Min**	**Max**	**Mean ± SD**	**Min**	**Max**
Sediments						
pH	6.99 ± 0.438*	6.22	8.00	7.75 ± 0.417	6.98	8.28
EC (mS cm^{-1})	−114 ± 44.4	−165	−44.4	−120 ± 45.2	−178	−52.7
Eh (mV)	2.46 ± 1.54	1.21	6.76	2.32 ± 2.52	0.750	8.20
TOM (%)	12.7 ± 4.44*	6.54	21.7	8.16 ± 2.33	5.26	12.4
Water						
Temp (°C)	29.4 ± 0.716*	28.2	30.3	30.5 ± 1.25	28.7	32.9
pH	6.97 ± 0.342*	6.10	7.39	7.48 ± 0.495	6.87	8.44
Salinity (%)	0.163 ± 0.0664	0.090	0.300	0.158 ± 0.0832	0.0700	0.310
DO (mg l^{-1})	7.90 ± 0.513	6.91	8.63	7.94 ± 0.232	7.43	8.29
BOD (mg l^{-1})	5.44 ± 2.47	2.69	9.84	6.98 ± 3.69	1.87	13.8
DOC (mg l^{-1})	9.31 ± 3.95	3.61	18.3	4.26 ± 2.04	0.510	6.49
TSS (g l^{-1})	0.0229 ± 0.00833*	0.0063	0.0369	0.0469 ± 0.0221	0.0173	0.0760
VSS (g l^{-1})	0.0143 ± 0.00317	0.0089	0.0189	0.0160 ± 0.00350	0.0119	0.0232
NH_3–N (mg l^{-1})	0.283 ± 0.239	0.0133	0.709	1.56 ± 2.15	0.0100	5.21
NO_2^-–N (mg l^{-1})	0.0310 ± 0.0687	0.000	0.244	0.113 ± 0.138	0.0041	0.423
NO_3^-–N (mg l^{-1})	0.0620 ± 0.112	0.0029	0.365	0.196 ± 0.210	0.0023	0.770
PO_4^{3-}–P (mg l^{-1})	0.0972 ± 0.0896*	0.0013	0.255	0.253 ± 0.136	0.0371	0.459
TP (mg l^{-1})	0.203 ± 0.143	0.000	0.492	0.372 ± 0.289	0.0072	0.868

Data from monthly sampling in July, August, September 1997 from landward side and seaward side, $n = 12$.
*One-way ANOVA results showed significant difference between landward and seaward sides ($p < 0.05$).

Table 3: Correlation matrices of physico-chemical variables of the water and sediments in *gei wais* at Mai Po Marshes (data from monthly sampling in July, August, September 1997 from landward side and seaward side, $n = 12$).

	pH (sediments)	Eh	EC	TOM	Temp	pH (water)	Sal	DO	BOD	DOC	TSS	VSS	NH4	NO2	NO3	PO4	TP
Landward (r)																	
pH (sediments)	1.000																
Eh	−0.141	1.000															
EC	0.014	−0.551	1.000														
TOM	−0.307	−0.175	−0.074	1.000													
Temp	0.229	0.132	−0.559	0.073	1.000												
pH (water)	−0.674*	0.100	0.191	0.035	−0.073	1.000											
Sal	−0.051	0.090	−0.399	−0.223	0.566	0.301	1.000										
DO	−0.128	−0.382	−0.396	0.401	0.192	−0.173	0.276	1.000									
BOD	0.097	−0.369	0.427	0.117	0.117	0.467	0.329	−0.229	1.000								
TOC	−0.273	−0.318	0.335	0.244	−0.405	0.365	−0.271	−0.006	0.350	1.000							
TSS	0.419	−0.388	0.290	−0.381	0.297	0.162	0.538	0.050	0.556	−0.230	1.000						
VSS	0.236	−0.477	0.431	−0.074	0.131	0.304	0.493	0.080	0.758**	0.195	0.836***	1.000					
NH4	−0.068	−0.146	0.540	−0.152	−0.871***	−0.099	−0.467	−0.126	−0.242	−0.045	−0.094	−0.113	1.000				
NO2	−0.559	−0.087	0.021	0.135	−0.479	0.064	−0.155	0.345	−0.403	−0.041	−0.389	−0.437	0.536	1.000			
NO3	−0.525	−0.025	−0.040	0.224	−0.319	0.105	−0.214	0.309	−0.402	−0.152	−0.371	−0.511	0.431	0.940***	1.000		
PO4	−0.095	0.527	−0.097	0.164	0.055	0.226	0.190	−0.332	0.332	0.174	−0.096	0.202	−0.264	−0.378	−0.451	1.000	
TP	−0.283	0.231	0.392	−0.124	−0.390	0.541	0.120	−0.403	0.443	0.320	0.171	0.433	0.196	−0.141	−0.279	0.741**	1.000
Seaward (r)																	
pH (sediments)	1.000																
Eh	−0.712**	1.000															
EC	−0.160	0.094	1.000														
TOM	−0.190	−0.072	0.104	1.000													
Temp	−0.180	0.322	0.242	−0.116	1.000												
pH (water)	−0.349	0.117	0.106	0.443	0.295	1.000											
Sal	−0.399	0.210	0.607*	0.517	0.585*	0.678*	1.000										
DO	0.144	−0.058	−0.003	0.262	0.317	0.482	0.253	1.000									
BOD	−0.267	0.444	0.186	−0.242	0.934***	0.112	0.416	0.309	1.000								
TOC	−0.062	0.253	0.399	−0.327	0.842***	0.036	0.375	0.168	0.781**	1.000							
TSS	0.000	0.068	−0.053	0.292	−0.111	−0.316	0.025	0.083	0.029	−0.151	1.000						
VSS	0.107	−0.047	−0.388	0.173	−0.473	−0.332	−0.343	−0.003	−0.306	−0.520	0.813**	1.000					
NH4	−0.404	0.383	0.803**	0.155	0.679*	0.423	0.830***	0.150	0.591*	0.637*	−0.208	−0.571	1.000				
NO2	−0.247	0.127	0.643**	0.295	0.559	0.132	0.712**	0.204	0.485	0.551	0.306	−0.257	0.648*	1.000			
NO3	−0.505	0.679*	0.070	0.075	0.793**	0.465	0.497	0.386	0.792**	0.562	−0.212	−0.413	0.613*	0.286	1.000		
PO4	−0.499	0.499	0.715**	0.236	0.326	0.330	0.625*	0.207	0.377	0.276	0.012	−0.355	0.767**	0.540	0.464	1.000	
TP	−0.527	0.440	0.731**	0.382	0.438	0.407	0.738**	0.161	0.404	0.359	−0.181	−0.526	0.886***	0.571	0.582*	0.903***	1.000

$p < 0.05$*; $p < 0.01$**; $p < 0.001$***.

degradation may dominate the interactions between the water and the sediments. Organic acids produced during organic matter decomposition contribute to low pH levels, while nutrients released from organic matter decomposition may promote algal growth in water, which may, in turn, lead to high water pH levels (Wetzel, 2001). The negative correlations ($p < 0.05$) between water pH and sediment pH may possibly result from organic matter degradation in the sediments. In the water column, heterotrophic bacteria promotion by organic matter (Wetzel, 2001) seems to explain the positive correlations ($p < 0.05$) between BOD and VSS. Moreover, the negative relationship ($p < 0.05$) between temperature and NH_3–N suggests that temperature affects NH_3–N metabolism (assimilation by bacteria or algae, or transformation by nitrobacteria) (Wetzel, 2001), that more NH_3–N was removed under higher water temperature.

At the seaward side, change in redox seems to be the major factor controlling the interactions between water and the sediments. It has been revealed that the oxidation of FeS_2 (and other sulphide minerals) to sulphate and NH_3–N to NO_3^- –N leads to the production of H^+ and low pH levels (Bourg & Loch, 1995). The negative correlations between pH and Eh in the sediments ($p < 0.01$) reflect this redox change in response to the electron-accepting reactions in aquatic systems. This result confirms the relationships between Eh and pH at the seaward side as previously reported (Lau, 2000). On the other hand, the close relationship between EC and salinity seems to imply possible sediment resuspension at the seaward side. Temperature has been shown to promote N and P release from the sediments in *gei wais* (Lau & Chu, 1999), while temperature is the key in governing microbial activities in the water (Wetzel, 2001). The close correlations among water temperature, organic matter (BOD, TOC) and nutrients (N, P) in the water possibly reflect the importance of temperature in the aquatic biological and physicochemical processes.

5.3.3. Relationships Between Heavy Metal Concentrations in the Sediments and Aquatic Physicochemical Properties in Gei Wais at Mai Po Marshes

The concentrations of total heavy metal and heavy metal speciations in the sediments are shown in Table 4. The distributions of heavy metals in different speciation fractions in the sediments from both locations are similar, except for the significant differences ($p < 0.05$) of Cr in the organic matter phase (Cr3) (landward: 2.81 $\mu g\, g^{-1}$; seaward: 1.44 $\mu g\, g^{-1}$) and Ni in the sulfide phase (Ni5) (landward: 16.6 $\mu g\, g^{-1}$; seaward: 19.7 $\mu g\, g^{-1}$). Cr has been shown to have greater humic acid sorption ability, compared with Cd, Zn and Ni (Sposito, 1986), but it has low mobility under acidic conditions (Reimann & De Caritat, 1998). It seems that more TOM input might provide more binding

Table 4: Summary of heavy metal concentrations ($\mu g\ g^{-1}$) (total and speciation) in the sediments in *gei wais* at Mai Po Marshes.

	Landward			Seaward		
	Mean ± SD	**Min**	**Max**	**Mean ± SD**	**Min**	**Max**
Cd						
$HNO_3 + H_2SO_4$ (total, Cd)	0.574 ± 0.441	0.000	1.27	0.992 ± 0.817	0.040	2.28
KNO_3 (exchangeable, Cd1)	0.078 ± 0.107	0.000	0.250	0.0508 ± 0.0769	0.000	0.200
KF (adsorbed, Cd2)	0.001 ± 0.003	0.000	0.010	0.0717 ± 0.139	0.000	0.420
$Na_4P_2O_7$ (org matter, Cd3)	0.136 ± 0.218	0.000	0.650	0.168 ± 0.252	0.000	0.800
Na_2EDTA (carbonates, Cd4)	0.057 ± 0.089	0.000	0.230	0.109 ± 0.155	0.000	0.500
HNO_3 (sulfides, Cd5)	0.085 ± 0.148	0.000	0.500	0.226 ± 0.369	0.000	1.30
Cr						
$HNO_3 + H_2SO_4$ (total, Cr)	20.5 ± 4.81	13.4	29.7	17.2 ± 3.85	10.9	22.2
KNO_3 (exchangeable, Cr1)	0.878 ± 0.520	0.230	2.01	0.670 ± 0.485	0.040	1.50
KF (adsorbed, Cr2)	1.23 ± 1.20	0.230	4.24	0.999 ± 0.842	0.200	2.80
$Na_4P_2O_7$ (org matter, Cr3)	2.81 ± 1.42*	0.600	4.68	1.44 ± 1.02	0.000	3.20
Na_2EDTA (carbonates, Cr4)	2.78 ± 1.15	1.23	5.20	2.58 ± 1.15	0.380	4.22
HNO_3 (sulfides, Cr5)	9.58 ± 3.21	3.83	16.9	8.03 ± 1.87	5.18	10.3
Cu						
$HNO_3 + H_2SO_4$ (total, Cu)	38.7 ± 12.1	24.8	58.6	34.0 ± 10.5	21.5	58.9
KNO_3 (exchangeable, Cu1)	0.804 ± 0.373	0.180	1.53	0.517 ± 0.317	0.000	0.990
KF (adsorbed, Cu2)	2.51 ± 1.02	1.16	4.41	2.01 ± 0.895	0.480	3.20
$Na_4P_2O_7$ (org matter, Cu3)	6.70 ± 4.40	2.78	16.3	5.03 ± 1.61	3.02	8.96
Na_2EDTA (carbonates, Cu4)	9.20 ± 4.74	2.81	20.5	7.23 ± 2.25	3.76	11.7
HNO_3 (sulfides, Cu5)	12.3 ± 3.77	6.29	19.8	11.5 ± 3.89	6.52	19.5

(*continued*)

Table 4: Continued.

	Landward			Seaward		
	Mean ± SD	**Min**	**Max**	**Mean ± SD**	**Min**	**Max**
Ni						
$HNO_3 + H_2SO_4$ (total, Ni)	36.2 ± 6.30	28.4	47.2	38.3 ± 8.30	21.9	50.3
KNO_3 (exchangeable, Ni1)	1.60 ± 1.33	0.080	4.33	1.95 ± 1.36	0.140	4.34
KF (adsorbed, Ni2)	2.53 ± 1.63	0.240	5.23	1.77 ± 1.24	0.600	5.01
$Na_4P_2O_7$ (org matter, Ni3)	3.71 ± 1.42	1.68	6.42	3.59 ± 1.91	0.640	6.63
Na_2EDTA (carbonates, Ni4)	5.05 ± 1.94	1.68	8.22	5.42 ± 1.95	2.58	9.71
HNO_3 (sulfides, Ni5)	16.6 ± 2.16*	13.3	20.1	19.7 ± 4.42	13.8	29.0
Pb						
$HNO_3 + H_2SO_4$ (total, Pb)	56.0 ± 9.58	36.8	67.6	51.8 ± 11.2	33.0	72.4
KNO_3 (exchangeable, Pb1)	1.90 ± 1.16	0.350	4.05	1.54 ± 1.04	0.200	3.20
KF (adsorbed, Pb2)	1.60 ± 1.00	0.320	3.83	1.99 ± 2.12	0.310	8.12
$Na_4P_2O_7$ (org matter, Pb3)	6.50 ± 4.39	0.920	13.5	4.95 ± 3.84	0.740	13.2
Na_2EDTA (carbonates, Pb4)	16.6 ± 7.66	2.76	29.7	21.2 ± 10.6	5.53	37.8
HNO_3 (sulfides, Pb5)	17.4 ± 6.84	5.75	26.5	13.9 ± 6.36	4.40	28.2
Zn						
$HNO_3 + H_2SO_4$ (total, Zn1)	129 ± 27.4	76.2	162	121 ± 27.2	74.1	174
KNO_3 (exchangeable, Zn2)	0.912 ± 0.775	0.030	2.36	0.500 ± 0.658	0.010	2.18
KF (adsorbed, Zn3)	1.13 ± 2.04	0.010	7.22	0.580 ± 0.776	0.000	2.60
$Na_4P_2O_7$ (org matter, Zn3)	16.4 ± 6.08	10.3	27.4	18.2 ± 9.91	8.68	39.4
Na_2EDTA (carbonates, Zn4)	23.7 ± 5.88	14.7	32.5	25.1 ± 5.91	15.5	39.1
HNO_3 (sulfides, Zn5)	48.0 ± 11.2	23.0	59.6	42.8 ± 9.12	26.1	60.7

Data from the monthly sampling in July, August and September 1997 from landward and seaward sides, $n = 12$.
*One-way ANOVA results showed significant difference between landward and seaward sides ($p < 0.05$).

sites for Cr, which led to a higher retention of Cr3 in the sediments at the landward side.

Percentage distributions of heavy metal speciation are shown in Fig. 2. Heavy metal sulfide phases (Me5) (landward: 31.9–47.2%; seaward: 28.0–52.5%) and carbonate phases (Me4) (landward: 13.7–29.1%; seaward: 14.1–39.3%) accounted for most the total metal contents of each heavy metal, except for Cd. The exchangeable (Me1) (landward: 0.767–3.69%; seaward: 0.406–5.00%) and adsorbed phases (Me2) (landward: 0.836–7.42%; seaward: 0.511–6.66%) had the least percentage. Large variations were detected for Cd distribution in different speciation fractions.

The relationships between total heavy metal contents and aquatic physicochemical properties were revealed through regression analysis (Fig. 3). Among the six heavy metals, total Cd and Pb in the sediments seem to be closely associated with aquatic environmental factors at both the landward and seaward sides, while no significant relationships ($p < 0.05$) were revealed for total Zn or Ni. The results demonstrate that high water pH levels caused high Pb and Cr precipitation and retention in the sediments. Cu may dissolute from the sediments by Cl^- complexation at high salinity (Lydersen et al., 2002), but the results of the present study showed that high Cu retention in the sediments occurred under high water salinity. Future studies are necessary in order to pinpoint the proper mechanisms. On the other hand, our result indicating that high suspended solids (TSS, VSS) increased Cd in the sediments also cannot be easily explained. Further investigations on the suspended particles are needed to elucidate these problems.

The relationships between the concentrations of fractionated speciations of heavy metals and aquatic physicochemical properties are presented in Table 5. At the landward side, organic matter and biological activities appear to influence heavy metal mobility. It is suggested that the negative regressions of temperature on the organic phases of Cu (Cu3) and Cd (Cd3) may result from reduced pH levels in the sediment due to increased organic matter degradation under higher temperature (Wetzel, 2001). Natural organic matter has been shown to compete for the adsorption of heavy metals over clay minerals (Schmitt et al., 2002). It is likely that the increase of particulate organic matter (indicated by TSS and VSS) led to the decreased Cu adsorption to clay minerals, which reduced the fractions of the adsorbed phase of Cu (Cu2). The negative correlations ($p < 0.01$) between the exchangeable phase of Cu (Cu1) and water salinity may possibly be explained by the effects of Cl^- complexation of Cu (Lydersen et al., 2002).

At the seaward side, variations in redox condition seem to play an important role in heavy metal mobility. The results indicate that adsorbed phases of heavy metals (Cr2, Cu2 and Zn2) are closely associated with water salinity, NH_3–N, TP, PO_4^{-3}–P, temperature, and sediment pH and EC (Table 5), which is in agreement

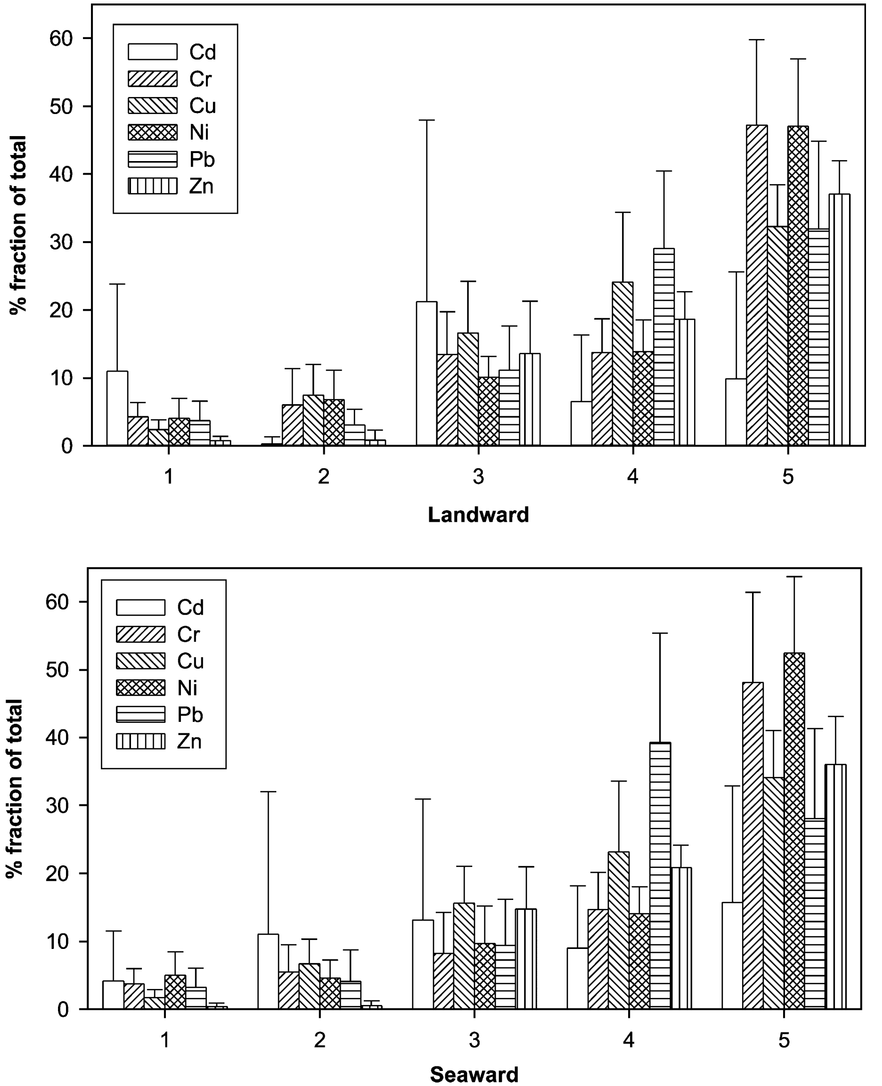
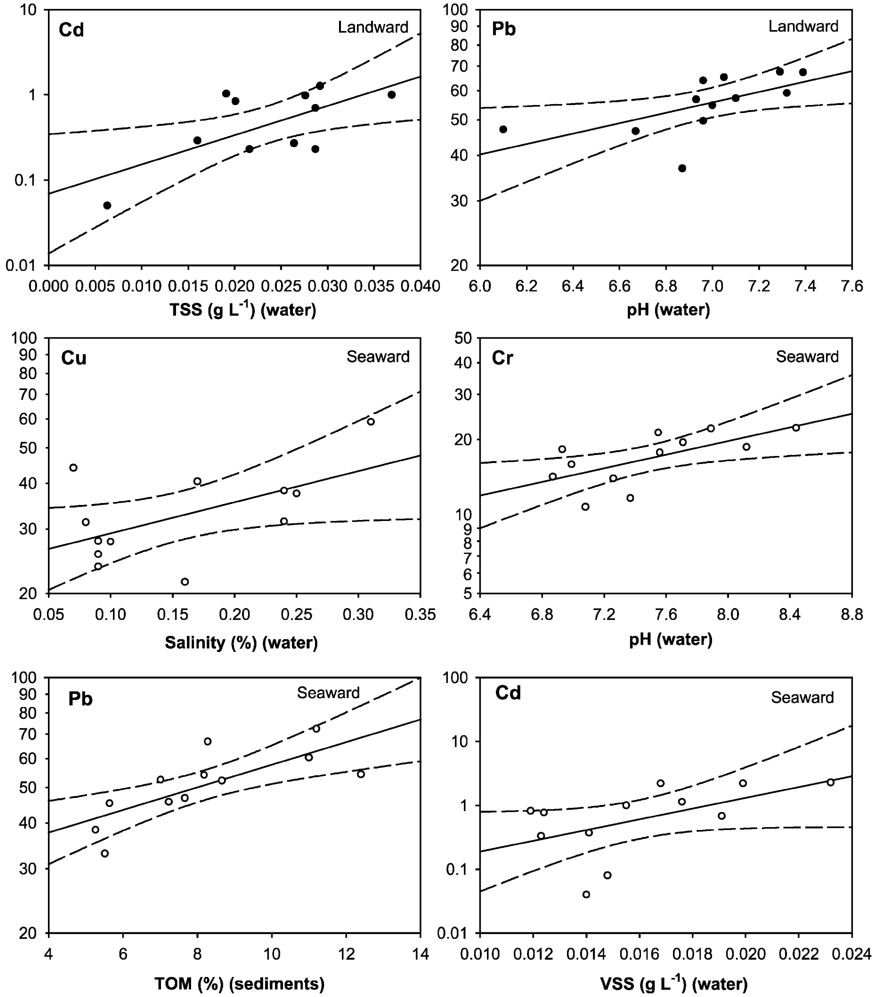


Figure 2: Percentage distribution of metal speciations relative to total metal concentrations season. Phases: 1—KNO_3 extracted, exchangeable; 2—KF extracted, absorbed; 3—$Na4P_2O_7$ extracted, organic matter bond; 4—Na_2EDTA extracted, carbonates; 5—HNO_3 extracted, sulfides). Errors bars represent one standard deviation about the mean.

with Yu et al. (2000) who found that Mn (hydr)oxides were responsible for the mobility of Cr, Cu and Zn in the sediments at Mai Po Marshes.

It is noticed that total Pb in the sediments of the seaward side was well predicted by TOM in the sediments ($p < 0.01$) (Fig. 3). Pb has much stronger affinity for



Log_{10} Cd = -1.16 + 34.4 TSS (r^2=0.436) Log_{10} Pb = 0.750 + 0.142 pH (r^2=0.374)

Log_{10} Cu = 1.38 + 0.853 Salinity (r^2=0.319) Log_{10} Cr = 0.227 + 0.134 pH (r^2=0.408)

Log_{10} Pb = 1.45 + 0.0309 TOM (r^2=0.563) Log_{10} Cd = -1.56 + 84.0 VSS (r^2=0.276)

Figure 3: Significant regression ($p < 0.05$) (95% confidence intervals) between heavy metals in the sediments ($\mu g\ g^{-1}$) and physico-chemical parameters at Mai Po Marshes.

sediment surface sorption and organic complexation over other heavy metals (Sposito, 1986; Calmano et al., 1994). It seems that during the sediment disturbance, Pb may have an advantage over other heavy metals in the formation of carbonated Pb, or complexes with the organic matter. Our heavy metal

Table 5: Significant linear regressions between heavy metal concentrations in the sediments (total and speciation) and physicochemical variables of the water and sediments in *gei wais* at Mai Po Marshes.

Landward					**Seaward**				
Variables	**Metals**	**Slopes**	R^2	p **value**	**Variables**	**Metals**	**Slopes**	R^2	p **value**
TSS	Cu2	−0.006	0.583	0.0039	EC (sediments)	Cu2	−2.074	0.541	0.0064
VSS	Cu1	−0.007	0.624	0.0022		Pb3	0.522	0.630	0.0021
	Cu2	−0.002	0.583	0.0038	Sal	Cu2	−0.075	0.646	0.0016
Temp	Cu3	−0.121	0.549	0.0058		Pb3	0.017	0.622	0.0023
Sal	Cu1	−0.135	0.574	0.0043		Pb4	0.006	0.529	0.0074
DO	Zn4	0.069	0.623	0.0023	NH4	Cu2	−2.211	0.845	0.0008
pH (sediments)	Cd3	1.489	0.550	0.0058		Pb3	0.500	0.794	0.0001
					TP	Cu2	−0.295	0.836	0.0001
						Pb3	0.057	0.575	0.0040
					PO4	Cu2	−0.127	0.694	0.0001
					Temp	Cr2	1.083	0.529	0.0074
					pH (sediments)	Zn2	−0.407	0.574	0.0043
					TOM (sediments)	Pb4	0.180	0.668	0.0012

Data from monthly sampling in July, August, September 1997 from landward and seaward sides, $n = 12$.

speciation results support this explanation, in that the organic matter complexed phase of Pb (Pb3) was well predicted by EC, salinity and NH_3–N and TP ($p < 0.01$) (Table 5). On the other hand, one-way ANOVA results showed that TOM was higher at the landward side ($p < 0.05$) than the seaward side, but there were no significant differences ($p < 0.05$) for total Pb (Liang & Wong, 2003). Compositions as well as concentrations of TOM may possibly affect Pb retention in the sediment. However, more studies are necessary to trace the source of the organic matter and the mechanisms regarding Pb behavior in the sediments.

5.4. Conclusions

There seems to be a spatial difference as to heavy metal mobility and its associations with the local aquatic biogeochemical processes at Mai Po Marshes. Organic matter decomposition seems to be the major process that determines heavy metal (Cd, Cr, Cu, Pb) mobility at the landward side, while redox condition dynamics at the seaward side appears to be the controlling factor for heavy metal (Cr, Cu, Zn) mobility. It is suggested that the uneven distribution of mangrove trees in *gei wais* may determine the organic matter (litter) input to the sediments and water temperature. Sediment resuspension and changes in redox conditions seem to be typical characteristics in estuarine systems, and sites closer to the coast seem to have more dramatic sediment resuspension and redox change in *gei wais* at Mai Po Marshes.

Organic matter appears to play an important role in heavy metal (Cd, Cr, Cu, Pb) mobility. Organic matter appears to complex heavy metals directly, decrease pH by releasing organic acid, and increase pH through releasing nutrients and stimulating algal bloom, while pH determines heavy metal retention in the sediments. The results imply that organic matter management at Mai Po Marshes should also consider organic dynamics associated heavy metal mobility.

At Mai Po Marshes, water quality and salinity seem to be important factors influencing heavy metal (Cu, Pb) mobility. Seasonal salinity and temperature gradients in *gei wais* at Mai Po Marshes were revealed in our previous study (Liang & Wong, 2003). Further studies on predicting temporal heavy metal mobility based on water quality and salinity are necessary in the future.

Suspended solid material was shown to well predict total Cd in the sediments in *gei wais*. While information on the composition of the suspended particles and its associations with the turbidity in the inter-tidal water has not been available, more investigations should be conducted.

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Developments in Ecosystems, volume 1
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Chapter 6

Biogeochemistry of Metals in the Rhizosphere of Wetland Plants — An Explanation for "Innate" Metal Tolerance?

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Abstract. Wetland plant rhizospheres are often aerobic and oxidized, while the bulk soils are anaerobic and chemically reduced. As a result metals tend to be immobile in the bulk soil, but are mobilized by plant-induced changes in the rhizosphere. This leads to enrichment of metals near the roots and enhanced availability for uptake by plants. It is proposed here that the apparent metal tolerance of wetland plants compared with dryland plants, without the development of separate metal-tolerant ecotypes, is due to the relatively high exposure of plant roots to metals under the soil conditions prevailing in wetlands.

6.1. General Biogeochemistry of Wetland Soils

One of the characteristics defining wetlands is the presence of hydric soils (Keddy, 2000; Mitsch & Gosselink, 2000). These soils tend to be less developed than dryland (as opposed to wetland) soils — they often lack a clear stratification into various horizons and are homogenous in texture — and are usually anaerobic and chemically reduced (Gambrell & Patrick, 1978). As a consequence, the dominant form of iron is the divalent ferrous iron, Fe^{2+}, while the prevailing form of sulfur is sulfide, S^{2-}. Pyrite and other iron–sulfur compounds may be formed, and metals such as zinc, cadmium and lead tend to precipitate as highly insoluble metal sulfides (Gambrell, 1994). The mobility of metals under such conditions is therefore low and this partly explains the successful application of wetlands for removal of metals from wastewater (Odum et al., 2000).

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6.2. The Rhizosphere of Wetland Plants

In order to live in the anaerobic soil conditions, wetland plants have developed a root morphology different from that of dryland plants. Many species have porous roots for the supply of oxygen for root respiration, often forming a specialized tissue known as aerenchyma or air tissue. This supply of oxygen is thought to be more than sufficient for root respiration and excess oxygen may leak into the rhizosphere. This is known as radial oxygen loss (ROL) (Armstrong & Armstrong, 1990).

The rhizosphere of wetland plants, therefore, can be aerobic, and the generally chemically reduced conditions of the bulk soil are reversed. Sulfur and iron are oxidized to form sulfate and the trivalent ferric iron, Fe^{3+}. As a consequence many wetland plants form a layer of ferric (oxy-) hydroxides on the root surface, also known as iron plaque (Mendelssohn et al., 1995).

6.3. Metal Mobility in the Rhizosphere of Wetland Plants

The presence of aerobic, oxidized conditions in the rhizosphere and the formation of iron plaque on wetland plant roots in an otherwise anaerobic, chemically reduced bulk soil has important consequences for the mobility of metals (Jacob & Otte, 2003). While bound to sulfides under the conditions prevailing in the bulk soil, metals such as zinc are highly immobile, because metal sulfides are generally insoluble. But, when these sulfides become oxidized to form soluble sulfates, the metals are mobilized. Consequently, wetland plants have been found to enhance porewater concentrations of metals (Fig. 1) (Wright & Otte, 1999). Rhizosphere oxidation may also lead to a decrease in pH, which could explain mobilization of metals independent from changes in redox status of the soil (Jacob & Otte, 2003), and Wright & Otte (1999) indeed found a reduction in pH coinciding with the mobilization of zinc due to the presence of plants.

Metals like zinc have a high affinity for adsorption and co-precipitation with iron (oxy-) hydroxides, and thus, metals mobilized in the rhizosphere diffuse towards iron plaque on wetland plant roots and are immobilized on or immediately adjacent to the root surface. The bulk soil thus acts as a source of metal while the iron plaque acts as a sink, creating a diffusion gradient driven by the oxidation processes in the rhizosphere. Compared to the bulk soil, iron plaque may be enriched in metals (Otte et al., 1989; Caçador and Vale, 2001), as may be the rhizosphere soil immediately surrounding the roots (Fig. 2) (Doyle & Otte, 1997).

Whether the iron plaque and rhizosphere are enriched depends on the metals and plant species involved (Crowder et al., 1987; Otte et al., 1989; Doyle & Otte,

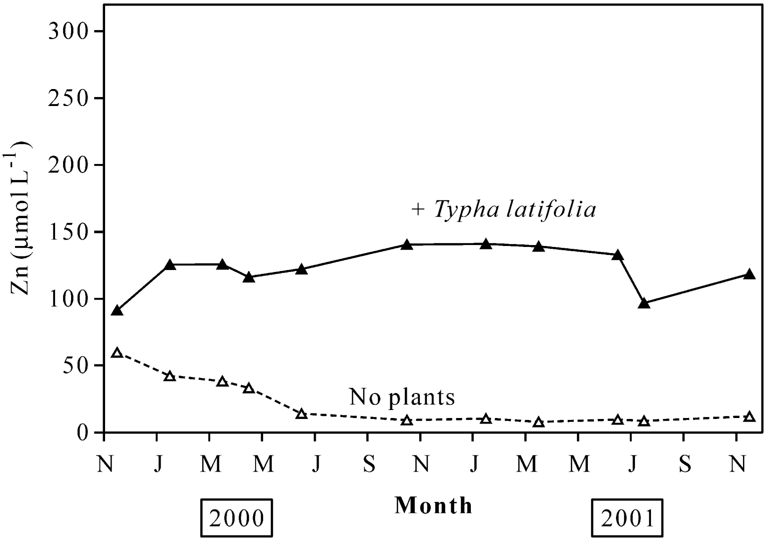

Figure 1: Concentrations of zinc in porewater of lead-zinc mine tailings from Tara Mines, Ireland, without (no plants) or planted with *Typha latifolia* during a 2-year period. For further details see Wright & Otte (1999).

1997; Ye et al., 1997a, 1998a; Caçador & Vale, 2001). However, metals adsorbed (as opposed to chemically bound) to iron plaque are probably still available for uptake by the plants, because plants can mobilize metals from adsorption sites directly or indirectly by exudation of protons or chelators (Marshner, 1988; Zhang, 1991). It is also likely that the exposure of plant roots to metals in solution at the root surface is higher if iron plaque is present than when it is not, because iron plaque accumulates metals relative to the bulk soil (Otte et al., 1989). This may be an important difference between wetland plants and dryland plants, as it may explain why wetland plants appear to be generally more tolerant to metals than dryland plants.

6.4. An Explanation for the Development of Innate Metal Tolerance in Wetland Plants?

Most studies on metal tolerance in plants have focused on dryland plants (Ernst, 1974; Verkleij & Schat, 1990), while metal tolerance in wetland plants has had very little attention. This recently changed with the increasing interest in using wetland cover for rehabilitation of mine wastes (Willianen et al., 1998; McCabe & Otte, 2000). Surprisingly, the few investigations into metal tolerance of wetland

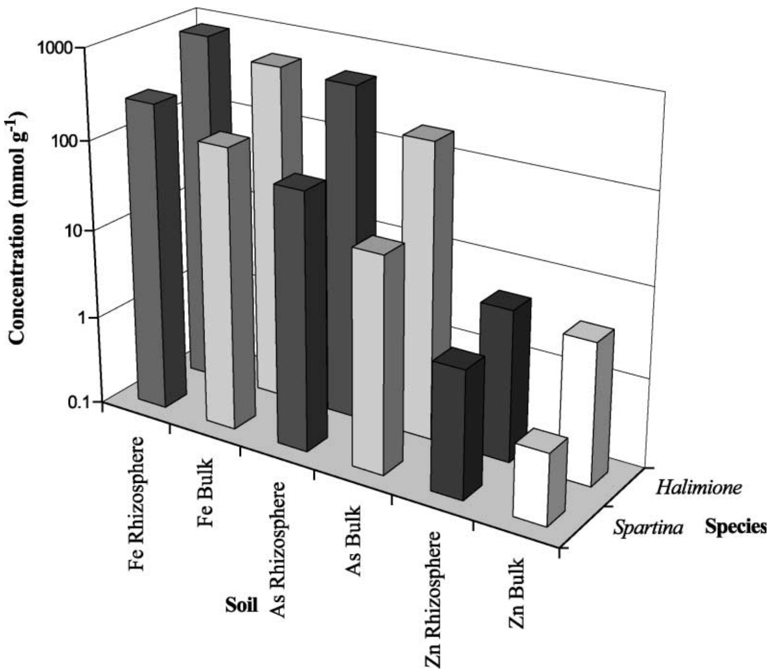

Figure 2: Concentrations of Fe, As and Zn in bulk soil and rhizospheres of *Spartina anglica* and *Halimione portulacoides* (after Doyle & Otte, 1997).

plants have been unable to identify differences in sensitivity between populations that had been exposed to various levels of metal concentrations in their respective habitats, as is normally found for dryland plants (Ernst, 1974; Verkleij & Schat, 1990). Populations from non-polluted and metal-enriched habitats showed similar growth performance in the presence of high concentrations of metals, suggesting that innate tolerance to metals is a common trait of wetland plants (McCabe et al., 2001). McNaughton et al. (1974) were the first to report on tolerance to zinc, cadmium and lead in *T. latifolia* and observed that this trait existed in this species without the development of specific metal-tolerant populations. Ten years later, Taylor & Crowder (1984) confirmed tolerance to copper and nickel in the same species. More recently, Ye et al. (1997b) confirmed the findings for *T. latifolia* and the same research team made similar observations in *Phragmites australis* (Ye et al., 1997c, 1998b). Then McCabe & Otte (2000) reported that populations of floating sweetgrass *Glyceria fluitans* that had not previously been exposed to zinc were capable of growing well in zinc-lead mine tailings containing highly elevated levels of metals. McCabe et al. (2001) and Moran & Otte (2004) found no differences in tolerance of *G. fluitans* to zinc when comparing a population from

Table 1: Tolerance index when exposed to 1000 μmol l^{-1} Zn (% response relative to control plants exposed to 2 μmol l^{-1}) for leaf length (ll), leaf number (ln), root length (rl) and survival rate (sr) of populations of four species of wetland plants, *Juncus articulatus*, *Juncus effusus*, *Eriophorum angustifolium* and *Glyceria fluitans* from non-contaminated (NC) and contaminated (C) locations in Europe.

Species	Population	NC/C	Origin	ll	ln	rl	sr
J. articulatus	Kippure	NC	Ireland	14	18	5	0
	Navan	C	Ireland	12	10	14	0
J. effusus	Cavan	NC	Ireland	23	33	31	40
	Glendalough	C	Ireland	22	34	4	10
E. angustifolium	Kippure	NC	Ireland	51	78	17	70
	Glendalough	C	Ireland	53	39	7	60
G. fluitans	Navan	C	Ireland	58	60	40	100
	Glendalough	C	Ireland	30	51	44	100
	Camborne	C	Cornwall	103	94	82	113
	Thisted	NC	Denmark	71	58	13	70
	Radostowo	NC	Poland	47	70	24	111

Plants were grown under the same conditions in a greenhouse in hydroponic culture ($n = 10$).

a zinc-lead tailings pond with one that had not been exposed to various elevated levels of zinc.

We have recent observations (Table 1) to suggest that there are wetland plant species, such as *Juncus articulatus*, that are less tolerant to zinc than the species mentioned above, but the evidence so far suggests that in contrast to dryland plants, metal tolerance in wetland plants is the rule rather than the exception. This cannot be explained by reduced uptake of metals — wetland plants do take up higher amounts of metals when exposed to elevated concentrations (Fig. 3).

The metal tolerance mechanisms in wetland plants, therefore, must be physiological and may have evolved in response to the biogeochemical conditions prevailing in the rhizosphere. It may be that over evolutionary periods of time wetland plants under natural, unpolluted conditions have consistently been exposed to higher concentrations of metals at the root surface than dryland plants.

This is supported by the following observations.

- Wetlands, particularly sedimentary systems such as salt marshes and estuarine floodplains, tend to act as sinks for metals and as a result the sediments of such wetlands tend to have higher metal concentrations than surrounding dryland habitats (Salomons & Förstner, 1984).

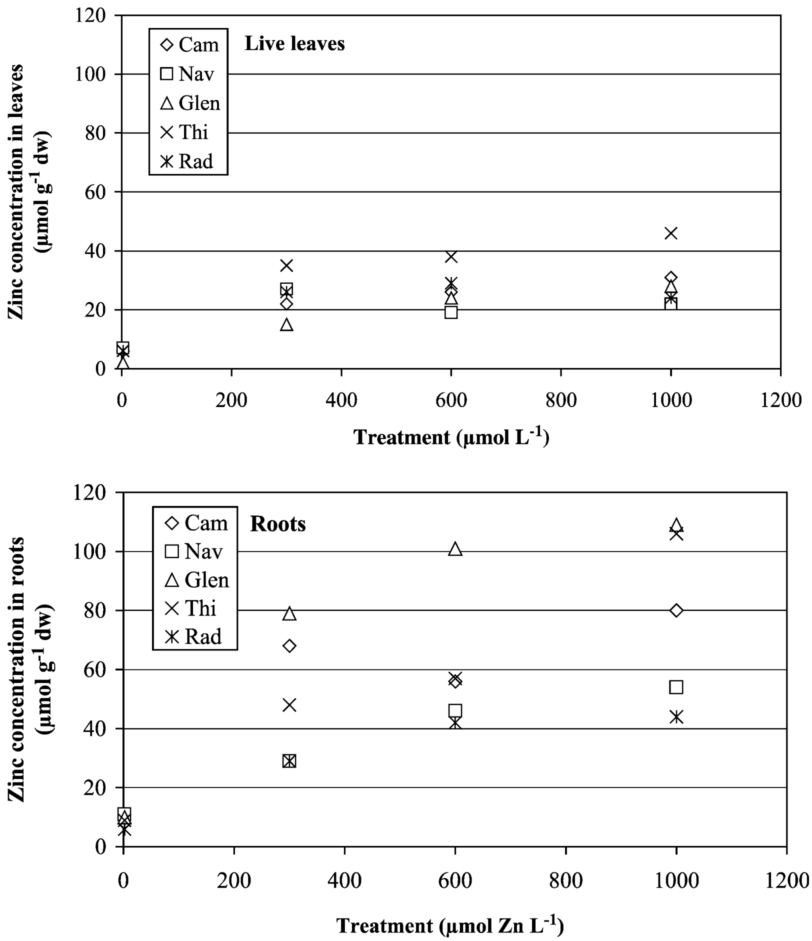

Figure 3: Mean zinc concentrations (µmol Zn g^{-1} dry weight) in live leaves (top) and roots (bottom) of four populations of *G. fluitans* from four populations: Camborne, Cornwall, UK (Cam), Glendalough, Ireland (Glen), Thisted, Denmark (Thi) and Radostowo, Poland (Rad), grown at 2, 300, 600 or 1000 µmol Zn l^{-1} for 84 days. Plants were grown under the same conditions in a greenhouse in hydroponic culture ($n = 10$).

- While the concentrations in the rhizospheres of dryland plants tend to have similar or lower metal concentrations than the bulk (or non-rhizosphere) soils (Youssef & Chino, 1989; Lorenz et al., 1997; McGrath et al., 1997; Luo et al., 2000; Wang et al., 2002), the situation tends to be the opposite in wetlands, as explained above (Fig. 2). In addition, wetland plants mobilize metals in the rhizosphere, even in mine tailings in which availability of zinc to plants is more than sufficient for growth (Fig. 1) (Wright & Otte, 1999).

We suggest here that this has led to higher uptake of metals by wetland plants compared to dryland plants and that this through evolution has led to the subsequent development of innate tolerance to metals in wetland plants. This hypothesis as well as the physiology and genetics underlying innate metal tolerance in wetland plants clearly need further investigation.

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Chapter 7

Mycotrophy and Its Significance in Wetland Ecology and Wetland Management

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Abstract. Rivers, marshes, creeks, and ponds are ecological habitats for plants adapted to withstand stress arising from waterlogging, anaerobiosis, and high salinity. Universal mycosymbionts like arbuscular mycorrhizal fungi may enhance the ecological adaptations of these plants to such environments. This chapter reviews the general mycorrhizal status of various life forms of aquatic macrophytes growing in such ecological habitats and the relationships of arbuscular mycorrhizae (AM) to redox potential in sediments and its P-status. Future studies will have to provide the data required to evaluate the effectiveness of mycorrhizal inoculation of wetland plants, including rice, in order to assess the significance of AM in wetland ecology and wetland management.

7.1. Introduction

Arbuscular mycorrhizal (AM) fungi, belonging to order Glomales, form symbiotic associations with roots of 80–90% of land plants in natural and agricultural ecosystems (Brundrett, 2002). Plants associated with these fungi benefit in the increased uptake of immobile nutrients like P, Zn, and Cu (Smith, 1980; Smith & Read, 1997; Jamal et al., 2002). In addition, AM mycorrhizal infection also leads to increased tolerance of plants to various soil environmental stresses like draught, waterlogging, and salinity (Khan, 1995). These associations represent a key factor in the below ground network which influences diversity and plant community structure (O'Conner et al., 2002), but we know very little about the enormous AM fungal diversity in soils and their properties and behavior in the soil environment (Khan, 2002a). However, not all plants are mycorrhizal and not all AM fungi benefit host plants under all growth conditions

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(Francis & Read, 1995). The presence of AM in aquatic and wetland plants suggests that they are ecologically significant, but their function is not well understood. The aim of this review is to assess the occurrence of arbuscular mycorrhizae (AM) in aquatic plants and their significance in wetland ecology and management.

7.2. Early History of Glomales

Glomales are one of the oldest group of fungi, older than land plants. The first land plants, Bryophytes, appeared in the Mid Ordovician to Early Silurian periods (430–476 million years old). The oldest fossil evidence of Bryophyte-like land plants, 100 million years ago in Early Devonian, had AM-like infections even before roots evolved. Phipps & Taylor (1996) reported morphological structures resembling modern arbuscular mycorrhizal fungi (AMF) in Triassic roots confirming evolution of mycorrhizal fungi by that time. The first land plants are most likely to have evolved from algae (Kenrick & Crane, 1997). No fossil records are available for the rootless freshwater Charophycean algae, which are the probable ancestors of land plants, to show if they were mycorrhizal. Mosses, liverworts, and hornworts often contain AMF structures like hyphae, vesicles, and arbuscules (Turnau et al., 1999; SchuBler, 2000). Fossil evidence of AM in the rhizomes of early vascular plants suggests that the origin of vascular terrestrial plants is likely to have been from their bryophyte-like ancestors (Edwards et al., 1998). Bryophytes represent a radically different way of doing things by evolving desiccation tolerance strategy for adaptation to life on land and, in species number, are the biggest group of green land plants (Proctor, 2000). Sphenophytes, Lycopodophytes and Pteridophytes are among the first land plants with roots which originated in mid-Devonian era, and AM associations are reported in these plants (for references, see Brundrett (2002)). Both living and Triassic fossil cycads had AMF in their roots. Members of Pinaceae have ectomycorrhizae. AM associations are ubiquitous in the living angiosperms (Newman & Reddell, 1987; Trappe, 1987), which probably arose in the early Cretaceous (Stewart & Rothwell, 1993; Taylor & Taylor, 1993). The phylogenetic relationships between origin and diversification of AM fungi and coincidence with vascular land plants was investigated by Simon et al. (1993) by sequencing ribosomal DNA genes (SS sequences) as a molecular clock to infer dates, from 12 Glomalean fungal species. The authors estimated the origin of AM-like fungi to be 353–462 million years ago, which is consistent with the hypothesis that AM were instrumental in the colonization of land by ancient plants. This hypothesis is also supported by the observation that AM can now be found worldwide in the angiosperms, gymnosperms as well as ferns, suggesting that its nature is ancestral.

7.3. Evolution of Roots

Early land plants faced selection pressure to increase surface area of their roots in soil. It has been suggested that roots evolved as habitats for mycorrhizal fungi: thus intracellular air spaces in the cortex of plants are for AM fungal hyphal growth. Large cortical cells in the roots are to accommodate AMF structures like hyphal coils, arbuscules and vesicles. The endodermis layer is for restricting AM fungal growth in vascular tissue of the roots, and the epidermis is for appresoria formation and root penetration by AMF hyphae. Long, slow-growing, long-lived, relatively thick roots are typical of plants with obligate AM associations. Plants with fine root systems exclude AMF and are non-mycorrhizal. Brundrett (2002) gave a detailed account of coevolution of roots and mycorrhizae in land plants, and assessed the recent evidence available from palaeobotanical and morphological studies and the analysis of DNA-based phylogenies. The author also discussed major evolutionary trends and the relative success of plants with different root types.

7.4. Roots of Aquatic Plants

The concept of aquatic plants adopted for this chapter is much wider and includes not only vascular plants living habitually in or on water but also some species living in semi-aquatic and terrestrial habitats subject to seasonal inundation due to flooding and or increase in height of the soil water table, after periods of heavy rain.

Roots and leaves of aquatic plants possess a well-developed system of air lacunae (Armstrong, 1979; Justin & Armstrong, 1987) that allow transport of CO_2 from sediment through root lacunae to leaves and transport O_2 in the opposite direction (Pedersen et al., 1995). Roots of wetland plants can adapt to anaerobic conditions via the development of these air lacunae (Armstrong et al., 1991). Mycorrhiza formation is the most important evolutionary factor determining presence or absence of lacunae in roots.

7.5. Mycotrophy of Aquatic Plants

In the past, less attention was given to the presence of those soil microorganisms, such as AM fungi, that are an integral part of wetland soil regimes and that may play an important role in the establishment and diversity of plants (DeAngelis et al., 1986). However, the phenomenon has received increased attention in the recent years (Cook & Lefor, 1998; Turner & Friese, 1998; Miller & Bever, 1999;

Thormann et al., 1999). The presence or absence of mycorrhizae in the plant species used in wetland restorations might be an important factor in the re-establishment of wetland plant associations.

7.5.1. Mycorrhizal Status

Information regarding the presence of AM in aquatic and wetland habitats is limited and often contradictory. AM in aquatic macrophytes were first recorded by Sondergaard & Laegaard (1977). This observation was subsequently supported by others (for references see review articles by Khan (1995) and Khan & Belik (1995)). Mycorrhizal status of the dominant families of aquatic plants, i.e. Cyperaceae, Juncaceae, and Typhaceae found in most wetlands, varies across habitats due to local environmental conditions. Because AM fungi are obligate aerobes, their occurrence and survival in the anoxic aquatic habitat, characteristic of wetland plants, was postulated to be the reason for non-mycorrhizal status of aquatic plants reported by earlier workers. But recently many reports of their occurrence in wetland habitats have appeared in the literature (Khan, 1993a; Turner & Friese, 1998; Miller & Bever, 1999). The survival of AM fungi in the aquatic plant root and near the plant root in the rhizosphere under anoxic conditions, may be due to the fungus obtaining oxygen directly from the root or as oxygen diffuses from the root into the rhizosphere.

Earlier reports regarding the mycorrhizal status of aquatic plants ranged from complete absence to reduced or temporary absence due to anaerobiosis, and recurring under aerated and drier conditions (Shuja et al., 1971). Khan (1974) reported absence of mycorrhizal infection in the roots of aquatic plants collected from diverse aquatic habitats in Pakistan but a re-examination of these hydrophytes by Hussain et al. (1994, 1995) found them to be mycorrhizal. It is possible that the endophyte may be existing in the reported non-mycorrhizal plants in the form of mycelia, not considered by the researchers as AM infection. The presence of arbuscules was used by earlier workers as a definition indicating the presence of AM fungi in plant roots (Malloh & Malloh, 1981), but the presence of vesicles alone in root cortex is now regarded as sufficient evidence for the presence of AM fungi (Thormann et al., 1999).

However, the presence of non-AM fungi like *Rhizoctonia* producing vesicle-like structures in plant roots may sometimes leads to misidentification (Dhillion, 1994). For example in addition to the AM fungal structures, Thormann et al. (1999) also reported the occurrence of a diverse assemblage of sterile dark-pigmented or hyaline, septate or non-septate, with or without clamp connections, fungal hyphae and swollen hyphal structures resembling vesicles of AM fungi: those structures occurred in and on the roots of plant species growing in peatlands along a bog-fen-marsh gradient in Southern Boreal Alberta, Canada. These fungi have also been found in great abundance in highly stressed ecosystems like peatlands (Thormann et al., 1999) and high altitude ecosystems (Malloh & Malloh, 1994; Treu et al., 1996). Turnau et al. (1999) suggested creating a new category of mycorrhizae to encompass these fungi. The role of these root endophytes and their ecological significance, if any, is unclear and needs further investigation (Jumpponen & Trappe, 1998).

AM infections were found in the young roots of trees growing on creek and river banks, in stationary or slowly flowing fresh water or brackish waters in swamps, creeks, drains and channels, and in seepage areas of New South Wales, Australia (Khan, 1991, 1993a, 1993b, 1993c). By contrast, the free-floating roots of these trees growing in water on the banks of the water bodies were non-mycorrhizal. Split root-system studies with rooted cuttings of Weeping Willow (*Salix babylonica*) confirmed that the free-floating roots became mycorrhizal when subjected to rooting sediments (Khan, 1991). The variations found in literature regarding mycorrhizal status are common in wet and waterlogged habitats, where conditions fluctuate on a seasonal or annual basis so as to favor or hinder mycorrhiza formation. Stevens & Peterson (1996) found higher AM colonization of roots of *Lythrum salicaria* growing in wetter areas in the field, and the reverse pattern in greenhouse experiments. Typically non-mycorrhizal plant species may become colonized under certain environmental conditions. Louise (1990) found that the number of mycorrhizal species increased from 0 to 87.5% as zones were sampled further from the sea. These and other examples suggest that soil saturation may be a limiting factor to colonization of wetland plant species that are typically non-mycorrhizal. However, the mechanism underlying these observations is still unclear and requires further research into the prevalence and role of AM in aquatic habitats.

Miller et al. (1999) studied the root hair morphology of 23 species of *Carex* (Cyperaceae), which are generally considered non-mycorrhizal, occurring in upland and wetland habitats, and found that certain species were typically mycorrhizal, other species were usually non-mycorrhizal and the mycorrhizal status of a third group of species depended upon the environment. The authors found lowest infection in the third group of *Carex* spp. in soils with high moisture and low pH, conditions typical of many wetland habitats.

Generally, AM infections noted in moist to wet habitats were found to be lacking arbuscules. A relationship between the characteristic AM infection pattern and soil moisture gradient was found in a study of a *Casuarina cunninghamiana* transect on a creek embankment (Khan, 1992), i.e. typical vesicles and arbuscules were found in roots from drier soils; there was a lack of arbuscules in relatively wet soils but large lipid-filled intracellular vesicles were present; and vesicles and arbuscules were absent in flooded creek bed where roots were associated with coenocytic intercellular hyphae containing abundant lipid droplets (Shuja et al., 1971; Khan, 1993a,b). These observations are consistent with those of Rickerl et al. (1994) and Stevens & Peterson (1996) who also reported high mycorrhization at the drier end of a moisture gradient. Miller (2000) also found lower AMF colonization in wetter sites for two wetland grasses. Thormann et al. (1999) reported no mycorrhizae in 11 species of herbaceous plants including sedges growing in three Alberta fens, but found AMF structures like arbuscules in a few roots and vesicles in many roots of woody fen plants. The authors attributed decreased arbuscular and increased vesicular frequency in the roots of mycorrhizal wetland plants, from early spring to summer and fall, to reported seasonality of VA-mycorrhizal formation in many plant species and ecosystems (Brundrett, 1991). Brown & Bledsoe (1996) and Sharma et al. (1998) have not documented arbuscules in the roots of aquatic plants they studied. Stenlund & Charvat (1994) reported that root colonization in aquatic plants is limited to only hyphae and vesicles. Miller & Sharitz (2000) found that flooding decreased the proportion of roots infected by AM fungi in wetland grasses, and that there was a direct relationship between flooding and mycorrhizal colonization. However, the fungi remained viable under flooded conditions in the roots of the wetland grasses, as shown by staining the roots with Nitroblue Tetrazolium (NBT)-succinate stain, suggesting that once mycorrhizal colonization has taken place, the fungus–root associations may endure prolonged exposure to flooding, depending upon the persistence of functioning and oxygenic roots (Miller, 2000).

Cornwell et al. (2001) studied mycorrhizal status of monocots and dicots growing in a ground-water-fed, low-P New York fen and found monocots generally non-mycorrhizal despite the fact that their roots showed a significantly higher percentage of air-filled root porosity. The authors also reported nine out of 10 dicot fen plant species to be mycorrhizal.

7.5.2. Plant Life Forms and Mycorrhizae

Mycorrhizae have been reported in ferns and angiosperms growing as emergent, free-floating or submerged (permanently or periodically) plants in aquatic environments (for references see review by Khan & Belik (1995)). Beck-Nielson & Madson (2001) have reported the presence of AM in five out of 25 emergent as well as submerged macrophyte spp. growing in streams. Typical infections with internal and external mycelia, and characteristic vesicles and arbuscules were found in submerged leaves of *Salvinia cucullate* (Bagyaraj et al., 1979). Belik & Khan (1992) reported well-established AM infection, mainly vesicles, in 38% of the root segments of *Ruppia polycarpa*, a submerged and surface-flowering plant growing in shallow waters at Prospective Reservoir, Sydney, Australia. A survey of 17 aquatic macrophytes in the Sydney area representing submerged, emergent, floating, and free-floating growth forms by Belik & Khan (1993a), showed that 35% of plants, including free-floating *Elatine gratilioides*, were mycorrhizal (Table 1). These findings established that AM fungi can infect permanently submerged plants under natural conditions.

It has been known for a long time that desiccation-tolerant liverworts and hornworts, which are the product of some 450 million years of evolution since their origin (Edwards et al., 1998), form AM-like infections (Stahl, 1949). These earlier observations are supported by many recent researchers (Rabatin, 1980; Duckett & Read, 1991; Turnau et al., 1999). SchuBler (2000) synthesized AM-like infections using Glomalean fungi (*Glomus claroideum*) and a bryophyte *Anthroceros punctatus* (L.) for the first time. These observations provide further support to the hypothesis that symbiotic association with AMF was a primary event during the evolution of land plants and that mycotrophy is the essential condition in land plants.

7.5.3. Relationships to Redox Potential

The reduced mycorrhizal infection reported in aquatic plants by various researchers may be due to the low availability of oxygen (Russell, 1977; Rabatin, 1980; Saif, 1981) to the roots under flooded conditions (Saif, 1983). Tanner & Clayton (1985) demonstrated that decreased redox potential was associated with significantly reduced AM infection. The redox potential in sediments of Danish streams with non-mycorrhizal specimens ranged from 54 to 280 mV and in sediments with mycorrhizal spp. from 250 to 530 mV, indicating that the redox potential of sediments might play a role in the development of AM associations. Belik & Khan (1993a) and Khan (1993a) found a correlation between mycorrhizal infection and redox potential. Mycorrhizae were absent or less frequent in roots of aquatic trees growing in reduced environment of waterlogged soils with low Eh values ($Eh < 150$ mV) than in well-oxidized terrestrial soils with higher Eh values ($Eh > 300$ mV). Khan (1992) reported AM infections in the roots of *Casuarina cunninghamiana* growing in a transect on a creek embankment in the marshy and periodically inundated soils, but the same plant spp. formed ectomycorrhizae as well as AM associations when growing in well drained and aerated soils with higher Eh values. Surface roots of aquatic plants, growing in swampy areas of Sydney with higher redox potentials, were found to be heavily mycorrhizal as compared to their counterparts in deep sediments with lower Eh values (Belik & Khan, 1993a).

Table 1: Mycorrhizal status of aquaphytes studied in Pakistan (1974 and 1995).

Hosts	Life form	% AM	AMF	References
Cyprus eleusinoides Kunth	An	0.0	Nil	Khan (1974)
Cyprus diformis L.	An	0.0	Nil	Khan (1974)
Eichornia crassipes (Mart.) Schlecht	F	0.0	Nil	Khan (1974)
Hydrilla verticillata (L.f) L.C. Rich	S	0.0	Nil	Khan (1974)
Juncus bufonius L.	An	0.0	Nil	Khan (1974)
Lemna polyrhiza L.	F	0.0	Nil	Khan (1974)
Nelumbium speciosum Wild.	F	0.0	Nil	Khan (1974)
Nymphaea lotus L.	F	0.0	Nil	Khan (1974)
Nasturtium officinale R.Br.	An	0.2	Nil	Khan (1974)
Populus euroamericana	An	EM/AM	H,V	Shuja et al. (1971)
Potamogeton crispus L.	S	100	H,V,A	Khan (1974)
Potamogeton indicus Roxb.	An	0.0	Nil	Khan (1974)
Phragmites karka (Ritz.) Trin. Ex Steu	An	0.0	Nil	Khan (1974)
Ranunculus aquatilis L.	An	0.0	Nil	Khan (1974)
Sagittaria guayanesis H. B. and Khan	An	0.0	Nil	Khan (1974)
Trapa bispinosa Roxb.	An	0.0	Nil	Khan (1974)
Typha angustifolia Bory and Chaub.	An	0.0	Nil	Khan (1974)
Vallisneria spiralis L.	An	0.0	Nil	Khan (1974)
Agrostis sp.	An	40.5	H,V,A	Hussain et al. (1994, 1995)
Conyzanthus sp.	W	100	H,V,A	Hussain et al. (1994, 1995)

Echornia crassipes L.	S,F	92	H,V,A	Hussain et al. (1994, 1995)
Hydrilla verticillata (L.F.) L.C. Rich	S	95.5	H,V,A	Hussain et al. (1994, 1995)
Lemna gibba L.	F	19.8	H,V,A	Hussain et al. (1994, 1995)
Myriophyllum spicatum L.	S	98.5	H,V,A	Hussain et al. (1994, 1995)
Mentha longifollia (L.) Huds.	W	100	H,A,V	Hussain et al. (1994, 1995)
Potamogeton nodosus Poir	S	100	H,A,V	Hussain et al. (1994, 1995)
Phragmites karka (Reitz.) Trin. ex Ste.	An	50	H,A,V	Hussain et al. (1994, 1995)
Setaria spp.	W	100	H,A,V	Hussain et al. (1994, 1995)
Scripus spp.	Am	100	H,V	Hussain et al. (1994, 1995)
Scirpus maritimus L.	Am,W	49.5	H	Hussain et al. (1994, 1995)
Vallisneria spiralis L.	S	100	H	Hussain et al. (1994, 1995)
Veronica beccabunga L.	W	100	H,A,V	Hussain et al. (1994, 1995)

Am = amphibious; S = submerged; F = free floating; An = anchored; W = wetland; H = hyphal; A = arbuscular; V = vesicular; EM = Ectomycorrhiza; AM = Arbuscular mycorrhiza. Highlighted plant species were found to be non-mycorrhizal in 1974 but mycorrhizal in 1995.

7.5.4. Relationship to Root Hairs

Baylis (1969) found that plants with few roots may become obligatory mycorrhizal in P-deficient soils. Root hair abundance and length were found to be negatively correlated with mycorrhizal dependence or benefit (Baylis, 1975; Schwelger et al., 1995). Belik & Khan (1992, 1993a,b) reported roots of submerged *Ruppia polycarpa* equipped with root hairs and heavily mycorrhizal. On the contrary, the roots of aquaphytic plants *Aster sabulatus* and *Elatine gratilioides* were devoid of root hairs and had a high AM colonization (Belik & Khan, 1993a,b) (Table 2), thus exhibiting the root hair–AM infection relationship. However, roots of *Ottelia ovalifolia* and *Pseudoraphis spenesceus* deviated from this relationship as they lacked root hairs but were non-mycorrhizal, and those of *Ludwigia peploides*, *Scripus validus*, *Triglochin procera*, *Typha orientalis* and *Schoenupluctus validus* had root hairs and no AM colonization (Belik & Khan, 1993b) (Table 2). Beck-Nielson & Madson (2001) found that submerged macrophytes collected from lakes and streams that were infected all lacked root hairs.

Miller et al. (1999) identified a unique root hair morphology with bulbous base, which was found to be associated with the non-mycorrhizal conditions of *Carex* species growing in waterlogged soils. The interspecific variations in the mycorrhizal status of *Carex* species was apparently connected with the morphology of the root hairs and the non-mycorrhizal condition, i.e. species with characteristic bulbous-based root hairs were non-mycorrhizal. Root hairs represent an adaptation to non-mycotrophy.

Clearly, however, the evidence presented above and in other recent publications indicates that the relationship between the mycorrhizal status of aquatic plants and root hairs must be re-evaluated.

7.5.5. Relationship to P-Status (Oligotrophic vs. Eutrophic Status)

The relationship between soil P concentrations and AM colonization has been reported to be negative in P-rich soils and positive in P-deficient soils (Khan, 1975). Under conditions of high P supply, plants accumulate and maintain high internal P concentrations, which reduces or inhibits AM infection (Smith & Read, 1997). A negative effect of elevated P levels on AM root colonization has been demonstrated for wetland macrophytes like *Lythrum salicaria* (White & Charvat, 1999), *Solidago patula* (Cornwell et al., 2001), and *Typha angustifolia* (Tang et al., 2001). The interpretation of AM infection and P uptake for wetland plants is, however, complicated by dual sources of available P (soil and water) and frequent wet and dry cycles. The lack of AM colonization in wetland plants may have been

Table 2: Presence (+) or absence (−) of root hairs, aerenchyma and AM associations of aquaphytes of Sydney region.

Hosts	Life form	Aerenchyma + or −	Hairs + or −	% AM	AMF
***Ruppia polycarpa*[a]**	**S**	**+**	**+**	**71**	**H,V,A**
***Ottelia ovalifolia*[b]**	**S**	**+**	**−**	**0**	**Nil**
Triglochin procera[c]	S	+	+	0	Nil
Typha orientalis[c]	An	+	+	0	Nil
Ludwigia peploides[c]	An	+	+	0	Nil
***Pseudoraphis spenescens*[b]**	**An**	**+**	**−**	**0**	**Nil**
Schoenuplectus validus[c]	An	+	+	0	Nil
Pasicaria decipiens	An	+	−	0	Nil
Scirpus validus[c]	An	+	+	0	Nil
***Elatine gratiliodes*[b]**	**F**	**+**	**−**	**30**	**H,V**
Paspalum paspaloides	An	+	+	48	H,V
Phragmites australis	An	+	+	55	H,A,V
***Aster sabulatus*[a]**	**An**	**+**	**−**	**62**	**H,V**
Sagittaria graminea	An	+	+	21	H

Am = amphibious; S = submerged; F = free floating; An = anchored; W = wetland; H = hyphal; A = arbuscular; V = vesicular; EM = Ectomycorrhiza; AM = Arbuscular mycorrhiza. From Belik & Khan (1992, 1993a,b). (1) Relationship between mycorrhizal status of aquatic plants, presence or absence of root hairs, and root parenchyma must be re-evaluated. (2) Data regarding the trophic status (P-availability) of aquatic systems missing and rarely examined in aquatic macrophytes studied. (3) No true relationship was found between the root aerenchyma and AM colonization in the aquatic macrophytes studied—needs further evaluation. Terrestrial plants with few or no root hairs are obligatory mycorrhizal in P-deficient soils (Baylis, 1975; Belik & Khan, 1993b; Schwelger et al., 1995). Root hair–AM infection relationships in aquatic plants ??

[a] Roots possessed root hairs and heavily mycorrhizal.

[b] Roots lacked root hairs but non-mycorrhizal.

[c] Roots possessed root hairs and non-mycorrhizal.

due to increased availability of P from both the soil and the water columns. The relationship between P availability and mycorrhizal status has been rarely examined in aquatic macrophytes.

Aquatic plant growth in many freshwater ecosystems is limited by the phosphorus source, which is often from agricultural soils (Schindter, 1977) and the population surrounding many lakes (Gibson, 1997). No relationship was found between AM infection and trophic status of the British lakes (Farmer, 1985). Clayton & Bagyaraj (1984), who observed AM infections in submerged plants from both oligotrophic and eutrophic lakes, also found no relationship between infection and trophic status of the lakes and the incidence of AM colonization. Rickerl et al. (1994), on the contrary, reported a high correlation between colonization levels and P for plants sampled from the dry regime and no correlation for plants sampled from the wet regime. Histological staining of *Vallisneria americana* roots revealed the widespread presence of arbuscules and thick-walled vesicles in its roots (Wigand & Stevenson, 1994). Lateral oxygen extrusion from the roots of aquatic macrophytes into the rhizospheres of submerged plants, which is regarded as necessary for the growth of aerobic AM fungi, may result in oxidation of iron to form distinct barriers to P uptake thus reducing the P availability in this zone. It may be that AM fungal hyphae facilitate P uptake by extending hyphae beyond this P-depletion zone in the rhizosphere and mobilizing adsorbed P in the rhizosphere; in return, submerged plants may promote the establishment of AM associations by lateral oxygen release (Wigand & Stevenson, 1994). Like their terrestrial counterparts, AM fungi in aquatic plants may also facilitate nutrient cycling, especially P. Wigand & Stevenson (1997) tested this hypothesis by using a submerged plant *Vallisneria americana* supplied with P^{33}-orthophosphate and recorded a greater incorporation of P^{33} into root cortex of the heavily infected roots (80%) of the mycorrhizal plants as compared to the fungicide-treated non-mycorrhizal plants. The authors also found that the level of P in the root tissue was over three times greater than in the sheath (iron hydroxide plaque) and six times greater than in the sediments surrounding the mycorrhizal plant, indicating that mycorrhizae facilitate P uptake of *V. americana* in the P-limited freshwater habitats. These observations suggest that AM fungal mediation was a major mechanism of facilitated P uptake and that the surrounding the mycorrhizal roots represent an additional mechanism of P assimilation by aquatic macrophytes.

In freshwater ecosystems, oxygen release into the rhizosphere can lead to the formation of mineral-hydroxide (Fe, Mn) sheath formation surrounding the roots (Wium-Andersen & Andersen, 1972; Wigand & Stevenson, 1997). This sheath results in P-absorption, making it less available for direct uptake (Jaynes & Carpenter, 1986). St-Cyr et al. (1993) also reported such a sheath for the submerged macrophyte, *Vallisneria americana*, which has also been described equipped with AM (Wigand & Stevenson, 1994). Sheath-thickness and composition vary among macrophytes due to variations in oxygenation, redox potentials and site (Wium-Andersen & Andersen, 1972; Sand-Jensen et al., 1982; Jaynes & Carpenter, 1986; Wigand & Stevenson, 1994).

Christensen & Wigand (1998) found that the freshwater *Lobelia dortmanna* L. with low tissue P contents showed significantly higher mycorrhizal infection than plants with high tissue P contents, indicating that AM associations may be important in P assimilation by rooted aquatic plants in habitats with low P availability. White & Charvat (1999) studied the effect of P availability on the AM status of an emergent aquatic, *Lythrum salicaria*, grown in hydroponic sand cultures at five P levels with or without AM fungal inoculum obtained from the wetland soil. The authors reported no infection in the control plants, the lowest or no AM infection in the roots of inoculated plants grown at the highest P levels, and the highest mycorrhizal colonization for plants grown at the lowest P levels. This pattern is consistent with that for terrestrial cereal crops (Khan, 1975). Further controlled studies mimicking natural conditions, as concluded by White & Charvat (1999), are needed.

Beck-Nielson & Madson (2001) reported AM infection in most submerged species sampled from oligotrophic lakes in Denmark, whereas no infection was found among submerged macrophytes sampled from eutrophic lakes. AM colonization may be an ecologically significant factor in soft water oligotrophic lakes, which have low P-availability in the sediments. Increased oxygenation and redox potential in the rhizosphere of plants growing in such ecosystems, lead to reduced P solubility (Christensen & Andersen, 1996; Christensen & Wigand, 1998). The presence of AM may ameliorate this problem by increasing surface area for nutrient uptake and acquiring P by hyphae.

Cornwell et al. (2001) found roots of *Typha latifolia* and *Carex lasiocarpa*, previously described as mycorrhizal in other wetland ecosystems, to be non-mycorrhizal in a phosphorus-poor fen in New York, suggesting that P availability may not be important in determining in which habitat these species form mycorrhizae. This further emphasizes the point that understanding the ecological functions of AM fungi in a given ecosystem requires further studies, requiring field surveys and manipulations in addition to green house experiments.

Many wetlands in the low-lying areas are very fertile because they receive nutrient inputs from surface runoffs and groundwater, while those on highly weathered and sandy or peat soils are nutrient poor. Few studies have considered fertility when attempting to relate mycorrhizal colonization to wetland characteristics. P input decreases a plant's dependency on AM fungi, regardless of the effects of flooding. Studies on effects of flooding on the role of AM fungi in wetlands are therefore likely to be more informative when conducted in less fertile areas or in P-rich habitats with low availability of P.

Miller & Sharitz (2000) suggested that AM fungal colonization has the potential to benefit grasses under wet conditions by increasing the size and P contents of wetland grasses under both dry and wet conditions, and that this is likely to increase with increased aeration from plant root or seasonal water drawdown. Tang et al. (2001) studied the relationship between P availability and AM colonization of *Typha angustifolia*, previously recorded by various workers to be non-mycorrhizal and provided clear evidence that AM fungi can colonize this aquaphytic plant under controlled experimental conditions as a function of P availability.

Recently, some reports revealed AM in rice cultivated under waterlogged conditions. Rice plants were shown to be positively responsive to colonization by indigenous AM fungi (Dhillion, 1992). The author found that the higher availability of P in the high P soil did not completely eliminate AM infection of roots in the three rice varieties studied. Positive responses to AM-inoculation have been demonstrated in wetland rice (Solaiman & Hirata, 1996, 1998). Purakayastha & Chhonker (2001) achieved a high intensity of root colonization in wetland rice inoculated with the AM fungus *Glomus etunicatum* by raising seedlings in the upland nursery under aerobic conditions. The authors found that the colonization persisted when the same seedlings were transplanted into pots under waterlogged conditions. Secilia & Bagyaraj (1994) have shown that, like terrestrial crops, different strains and species of AM fungi differed in their ability to confer growth benefits to rice plants. Further studies are needed to evaluate the effectiveness of mycorrhizal inoculation of this wetland crop.

7.6. Significance of Mycotrophy in Wetland Ecology and Management

The literature reviewed above suggests that the occurrence of AM in aquatic hydrophytes is common and is associated with fluctuating water, nutrients, and oxygen conditions. However, the consequences of severity and duration of hypoxia, and the anatomical and physiological adaptive features of hydrophytes are not yet fully understood in relation to VA mycorrhizae. Since each AM fungal species and isolate has specific ecological requirements, the screening of their diversity in aquatic environment is necessary in order to select the most superior, effective and efficient isolate for successful introduction into a habitat (Khan, 2002a,b). Several areas of inoculum production and application technology for wetland management merit further investigations.

Are AM fungi performing similar functions in wetlands as they are in drier habitats? Recent research by various workers reviewed in this article suggests that AM fungi may be advantageous to wetland plants: (1) during periods of low soil moisture associated with seasonal hydrology, that leaves them with reduced water levels or completely dry for short or long durations; and (2) in wetlands with low P. Details of the ecology of AM fungi in aquatic environments are not well understood and documented. Knowledge of interactions between AMF in submerged sediments and the mycorrhizospheres of fibrous plant roots can lead to improved management of wetlands and may be applicable to wetland plant establishment and later management.

The habitat sampled by various authors, listed above, ranged from mineral-rich groundwater-fed wetlands (such as fens and wet meadows) where soils are saturated during much of the year, to marshes with stagnate water, and to downward gradient of water as in may seasonal wetlands. There are many biochemical differences between these habitats, which affect the mycorrhizal status of their macrophytes. The plants, growing in the organic soils of groundwater-fed wetlands surveyed by Turner et al. (2000) for AM associations, were subjected to habitats with lower bioavailability of P contents than marshes and other similar nutrient accumulating wetlands. The authors found up to 61% AM colonization levels in 90% of the plant species sampled, and proposed that in wetlands with low P availability AM may tend to be more numerous than in wetlands that act as a sink for nutrients, and that plants may be more dependent on AM fungi for nutrient uptake under such conditions. The authors proposed that while preparing a site for restoration or creation of a new wetland, importance of AM should be considered as a contributing factor to the success of the project, especially if nutrient availability is low.

Presence of AM in rice plants is interesting but requires further studies with reference to AMF diversity and functionality in order to evaluate the effectiveness of mycorrhizal inoculation in this wetland crop.

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Chapter 8

Assessment of Risks to the Mai Po/Inner Deep Bay Ramsar Site due to Environmental Contaminants

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Abstract. Within the Mai Po/Inner Deep Bay Ramsar site of Hong Kong, relatively high levels of trace metals and persistent organic contaminants were found in the sediments and biota. Levels of polychlorinated dibenzodioxins and dibenzofurans in the sediments from the Mai Po Marshes Nature Reserve were measured. When expressed on a toxicity equivalency basis, the concentrations ranged from 11 to 16 pg I-TEQ/g dry wt. Ecological risks to the Mai Po/Inner Deep Bay area due to environmental contaminants were assessed and some recommendations are made to elucidate the fate and environmental effects of these contaminants.

8.1. Background

Pollution of marine waters reduces the quantity and quality of available marine resources and, where toxic pollutants are involved, may also have a deleterious impact on human health via the consumption of contaminated seafoods. Due to its proximity to one of the most densely populated areas and busiest ports in the world, the coastal environment of Hong Kong has come under severe stress. Discharges of largely untreated domestic and industrial wastewater and the disposal of contaminated mud into Hong Kong's coastal waters have resulted in high levels of toxic contaminants in the water column, biota and bottom sediment (Wu, 1988; Blackmore, 1998; Connell et al., 1998a). In addition, the multi-billion dollar Harbour Area Treatment Scheme (HATS) (currently under construction) will discharge the wastewater from over 3 million people into surrounding waters, even though the impact of this release on the coastal environment is largely unknown. Previous studies of toxic contaminants in Hong Kong have tended to focus on monitoring heavy metal concentrations, while less attention has been given to persistent organic contaminants (POC).

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In a recent review, Connell et al. (1998a) investigated the occurrence of POC, based upon previous monitoring at a total of 66 sites in Hong Kong waters over the past 10 years. They concluded that the contaminants concerned, including petroleum hydrocarbons (PHCs), polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs) and organochlorine (OC) pesticides, were likely derived from a combination of industrial discharges, stormwater runoff, sewage and combustion (Connell et al., 1998a; and references therein). Further assessment of risks associated with the above POC in the Victoria Harbour was also performed using the routine monitoring data (1996–1999) from the Hong Kong Environmental Protection Department. Fugacity modeling was employed to calculate the aqueous and biotic concentrations (from POC concentrations in sediments), and the estimated levels were then compared with relevant guidelines. Based on this analysis, it was concluded that POC (e.g. PAHs and PCBs) were at levels which might pose a risk to local marine ecosystems and seafood consumers (Connell et al., 1998b). They also noted that the risks posed by other contaminants, including total alkanes, non-aromatic hydrocarbons, linear alkyl benzenes and chlorohydrocarbons, were likely to be additive to those of other POC present, and required further investigation to elucidate their distribution and ubiquity in local waters.

8.2. The Mai Po and Inner Deep Bay Ramsar Site

The Mai Po and Inner Deep Bay wetland, the largest remaining system of its type in Hong Kong, was listed as a Wetland of International Importance under the Ramsar Convention in September 1995 (Fig. 1). The Mai Po Marshes, occupying an important area for biodiversity conservation within the Ramsar site, were originally designated as a Site of Special Scientific Interest in 1976, and have been managed as a nature reserve by the World Wide Fund for Nature Hong Kong since 1984. Based on the bird list of Mai Po, records accumulated over several decades, there are over 380 bird species recorded in the area. Specifically, Mai Po supports the second largest known group in the world of Saunders' gulls (*Larus saundersi*) and one fourth of the world population of the Black-faced spoonbill (*Platalea minor*). Indeed, records from regular ringing exercises carried out at the Mai Po Marshes since 1979 show that the Deep Bay and Mai Po area is becoming increasingly important as a staging ground for migrant birds, especially shorebirds, to build up fat stores to provide fuel for the next stage of their migration.

There is increasing evidence that the high levels of environmental contaminants in the marine sediments around the Deep Bay area may be attributed largely to contamination sources in the Chinese mainland (Richardson & Zheng, 1999;

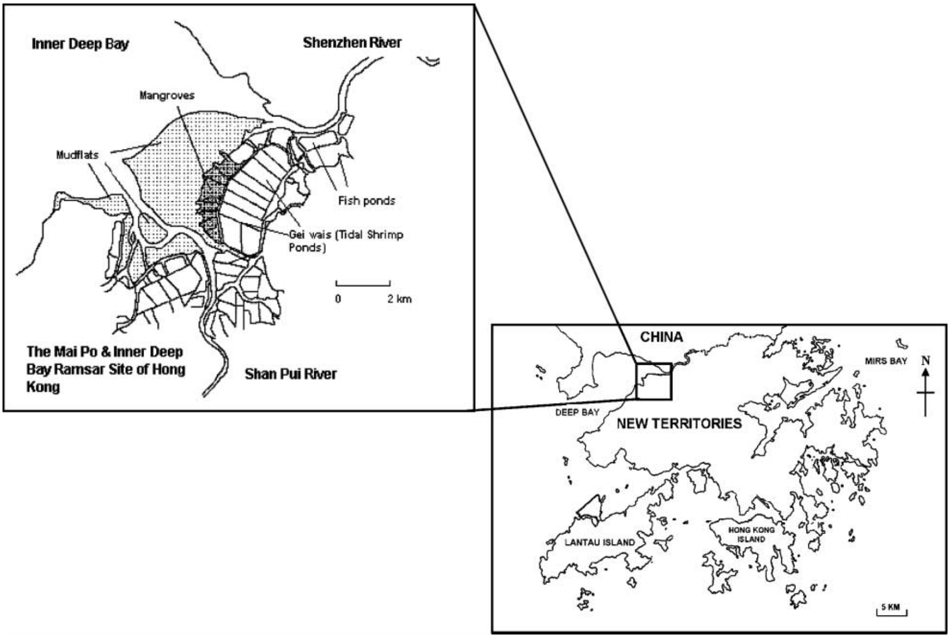

Figure 1: The Mai Po and Inner Deep Bay Ramsar Site of Hong Kong.

Richardson et al., 2000). The lower reaches of the Pearl River, receiving 2 million tonnes of various types of wastes and wastewater annually, are heavily polluted by domestic, industrial and livestock waste, and also by agrochemicals. Indeed, the annual discharge of the Pearl River (340 billion m^3), with an average sediment

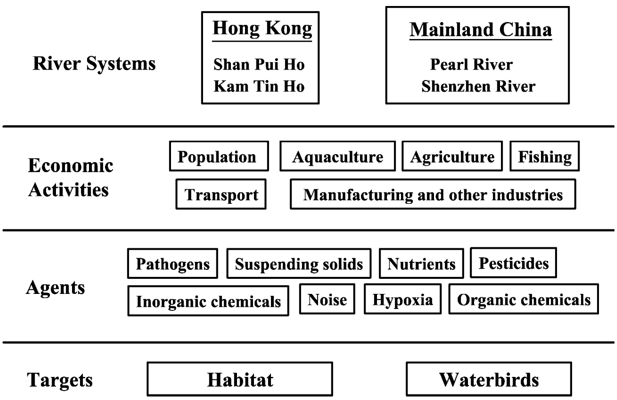

Figure 2: A simplified environmental profile of the Mai Po/Inner Deep Bay Ramsar Site showing major river systems, economic activities, and potential agents and targets.

load of 84 million tonnes, accounts for over 85% of the total nutrient input to the local waters of Hong Kong. This, together with inputs from the nearby polluted Shenzhen River and local streams, have been of major concern to maintenance of the long-term sustainability of the Mai Po and Inner Deep Bay ecosystem. A simplified environmental profile of the Mai Po/Inner Deep Bay area is given in Fig. 2. This report attempts to summarize results from recent studies on environmental contaminants in the Mai Po and Inner Deep Bay area, and assess the ecological risks to the Ramsar site due to these contaminants.

8.3. Levels and Risks of Environmental Contaminants in Sediments

Trace metals in Hong Kong's coastal waters, sediments and biota have been the subject of a number of studies spanning the 1970s to the 1990s, and the resultant data have been extensively reviewed by Blackmore (1998). Despite the existence of a large dataset on sediment trace metal contents and some information on the tissue metal content of certain marine organisms, their ecotoxicological significance has not been thoroughly investigated. A study, commissioned by the Agriculture and Fisheries Department (now Agriculture, Fisheries and Conservation Department, AFCD), examined the vertical profile of key inorganic (Cd, Cr, Cu, Hg, Ni, Pb, Zn, Fe and Mn) pollutants in porewater at selected sampling stations on the Mai Po mudflat and mangrove areas (Lam & Lam, 2000). Concentrations of cadmium, chromium, copper, lead and zinc in the sediment porewater of the Mai Po mudflat were found to vary in the range <1.1 ng/l–6.2 μg/l, <0.52 ng/l–20.8 μg/l, 0.6 ng/l–212.1 μg/l, <2.1 ng/l–24.0 μg/l, <0.7 ng/l–1,151.0 μg/l, respectively. For [Cd], [Cu], [Pb] and [Zn], remobilisation of trace metals from sediment to porewater occurred in the oxic layer close to the sediment–water interface, while the remobilisation of [Cr] occurred at the oxic/sub-oxic boundary. Metals in the porewater may be absorbed and accumulate in the tissues of benthic organisms, and may have an influence on predatory birds via the food chain.

Within the Ramsar area, total PAH concentrations in sediments collected from within the mangrove zone ranged between 666 and 1,042 ng/g dry weight. Zheng & Richardson (1999) reported a slightly lower level of total PAHs (558 ng/g) in sediments collected on a mudflat off Tsim Bei Tsui, Deep Bay. Naphthalene contents were high in the mangrove sediments, but mostly undetectable on the mudflat. There was a marked decline in total PAH levels from the landward end (about 800 ng/g near the mangrove fringe) towards the seaward end (212–355 ng/g) of the mudflat. The high total PAH concentrations in mudflat sediment collected near the mangrove fringe were partly attributed to the relatively high levels of specific PAH congeners, including benzo[a]pyrene and dibenzo(1,2,5,6)anthracene, which are potential carcinogens. In general, levels of total PHCs ranged from 267 to 363 μg/g dry weight, and did not show marked variations across different sampling stations in the Mai Po Nature Reserve. Total PCB concentrations in the Mai Po sediments were comparable to levels recorded elsewhere in Hong Kong (Richardson & Zheng, 1999). Levels of individual chlorinated pesticides were relatively stable (generally between 1 and 10 ng/g dry weight) across the mangrove and the open mudflat, except towards the seaward end where there was an apparent increase in chlorinated pesticide levels (Zheng et al., 2000). There is now increasing evidence that the high levels of OC pesticides in the marine sediments around the Deep Bay area, including compounds that have been banned in Hong Kong (e.g. DDT), are largely attributed to contamination sources in the Chinese mainland.

Tam et al. (2001) found that the PAH profiles of surface sediment samples from the Mai Po mangrove swamps had higher percentages of low-molecular-weight PAHs, and suggested that the PAHs might have originated from petrogenic sources. Zheng et al. (2002) analysed sediment cores (0–35 cm below surface) retrieved from 12 sampling stations in the Mai Po Marshes Nature Reserve of Hong Kong in 1999, and the vertical profiles of 15 priority PAHs in each sediment core were determined. On the mudflats, vertical profiles of the PAHs were quite uniform. At the fringe of the Mai Po mangroves, significantly higher concentrations of all PAHs were observed at the upper 0 to −8 cm layer. No significant difference in the distribution patterns of the 15 priority PAHs in summer and winter was observed, indicating that distribution of PAHs in the sediment of the Mai Po Marshes was not sensitive to sub-tropical climatic changes of the region. Two PAH isomer ratios, [Phenanthrene]/([Phenanthrene] + [Anthracene]) and [Pyrene]/([Pyrene] + [Fluoranthene]), were used to identify potential sources of PAH contamination in the wetland. The results suggested that local deposition might be a more important source than long-range atmospheric transportation.

There have been very few studies on PCDD and PCDF in Hong Kong. Müller et al. (2002) determined the concentrations of 2,3,7,8-substituted PCDDs and PCDFs in 14 sediment samples collected from four sampling stations in the Mai Po Marshes Nature Reserve and from another six sites in Victoria Harbour and along the Hong Kong coastline. Concentrations of PCDD/Fs were detected in all samples collected in the Mai Po Marshes and other sites from Hong Kong. In the Mai Po Marshes the mean concentrations of the $\sum$2,3,7,8-PCDD/Fs ranged from 5,000 to 6,900 pg/g dry wt in surface sediments and from 5,300 to 7,000 pg/g dry wt in sediments from various core depths. When expressed on a toxicity equivalency basis the concentrations in the sediments from the Mai Po Marshes ranged from 11 to 16 pg I-TEQ/g dry wt. The PCDD/F congener profiles in all samples are dominated by OCDD which contributed 94–97% to the ∑2,3,7,8-PCDD/F concentration in the samples. PCDD/F levels and congener profiles in the samples from the Mai Po Marshes Nature Reserve suggest that these contaminants have nonanthropogenic sources.

8.4. Levels and Risks of Environmental Contaminants in Biota

Liang et al. (1999) measured the PCB levels in fish and shrimps collected from tidal ponds at the Mai Po Marshes Nature Reserve, and concluded that PCB levels in the Grey Mullet exceeded the guideline value for human consumption, but the PCB concentrations in the aquatic organisms posed no hazard to fish-eating birds. In 2000, AFCD commissioned a study on the tissue contaminant levels of selected faunal groups in the Mai Po and Inner Deep Bay Ramsar Site (Lam & Lam, 2001). The study was mainly designed to provide data for an assessment of risks to predatory waterbirds due to consumption of potential food items in the area. A list of the eight faunal groups/species selected for analysis is given in Table 1. Tissue concentrations of trace metals and persistent organic pollutants (PHCs, OC pesticides, PCBs and PAHs) were analysed, and the results are summarised in Tables 2 and 3.

The potential risks of environmental contaminants to wildlife in Mai Po were assessed by comparing environmental conditions (e.g. environmental concentrations of toxic chemicals) with threshold values likely to cause adverse effects in the targets under consideration. In the type of risk assessment undertaken in this project, this was made explicit as a risk quotient (RQ) that is the ratio of an environmental concentration (either predicted (PEC) or measured (MEC)) with

Table 1: The eight faunal groups/species selected for analysis of persistent toxic substances in the Mai Po Marshes Nature Reserve.

Faunal group	**Common name**	**Scientific names**	**Sampling site**
Shrimps	*Gei wai* shrimp	*Metapenaeus ensis*	*Gei wais*
		Exopalaemon styliferus	*Gei wais*
Fish	Grey Mullet	*Mugil cephalus*	*Gei wais*
	Tilapia	*Tilapia mossambicus*	*Gei wais*
	Mudskipper	*Boleophthalmus pectinirostris*	Mudflat
Crabs	Fiddler crab	*Uca arcuata*	Mangroves
		Varuna litterata	*Gei wais*
Polychaetes			Mudflat

Table 2: Summary of mean tissue levels of metals in the eight faunal groups.

	Tissue concentration (μg/g)							
	Zn	Fe	Mn	Cu	Hg	Cd	Cr	Pb
Boleophthalmus pectinirostris	84.61	197.78	12.56	2.08	0.02	0.012	1.104	1.56
	(7.06)	(60.30)	(4.32)	(0.37)	(0.03)	(0.009)	(0.35)	(0.88)
Metapenaeus ensis	55.16	132.85	80.91	48.00	0.02	0.017	0.748	0.204
	(5.43)	(38.36)	(23.89)	(6.19)	(0.001)	(0.007)	(0.218)	(0.0082)
Uca arcuata	88.75	2340.20	385.40	70.11	0.07	0.49	2.555	5.985
	(8.48)	(732.67)	(182.74)	(19.65)	(0.03)	(0.22)	(0.93)	(1.923)
Mugil cephalus	75.50	969.84	67.63	3.24	0.0084	0.0071	1.326	0.58
	(10.169)	(1030.35)	(24.10)	(1.65)	(0.0036)	(0.0026)	(1.314)	(0.44)
Tilapia mossambicus	100.03	1480.00	122.25	5.76	0.015	0.010	1.40	0.74
	(12.42)	(840.00)	(81.02)	(1.37)	(0.0034)	(0.0038)	(0.43)	(0.38)
Exopalaemon styliferus	63.41	75.27	110.15	55.16	0.0065	0.0052	0.17	0.044
	(5.73)	(19.95)	(34.97)	(2.51)	(0.0021)	(0.0024)	(0.068)	(0.012)
Varuna litterata	75.24	947.46	371.99	54.98	0.016	0.017	0.25	0.16
	(9.61)	(112.04)	(92.25)	(7.17)	(0.0054)	(0.018)	(0.083)	(0.033)
Polychaetes	129.89	2795.35	38.36	20.17	0.0094	0.018	1.85	1.48
	(3.92)	(40.27)	(0.69)	(0.41)	(0.0038)	(0.0049)	(0.11)	(0.052)

Standard deviations are given in parentheses.

Table 3: Summary of mean tissue levels of persistent organic pollutants in the eight faunal groups.

	Tissue concentration (μg/g)							
	Total PAHs	Total PHCs	Total HCHs	Heptachlor	Chlordane	DDE	DDT	Total PCBs
Boleophthalmus pectinirostris	0.854	175.35	0.021	0.007	0.082	0.063	0.015	0.77
	(0.58)	(30.64)	(0.014)	(0.0026)	(0.11)	(0.040)	(0.022)	(0.30)
Metapenaeus ensis	0.93	91.25	0.015	0.0012	0.0015	0.014	0.0010	0.069
	(0.31)	(36.62)	(0.024)	(0.00038)	(0.00057)	(0.0074)	(0.0006)	(0.036)
Uca arcuata	0.75	151.84	0.0037	0.0039	0.00055	0.00048	0.0010	0.011
	(0.33)	(62.53)	(0.0051)	(0.0024)	(0.00021)	(0.00038)	(0.00044)	(0.009)
Mugil cephalus	0.98	574.35	0.0060	0.0018	0.0020	0.0011	0.0013	0.200
	(0.19)	(110.11)	(0.0014)	(0.0011)	(0.0011)	(0.0005)	(0.0007)	(0.050)
Tilapia mossambicus	1.443	672.0	0.019	0.0009	0.0041	0.0018	0.0028	0.086
	(0.214)	(84.6)	(0.022)	(0.0007)	(0.0010)	(0.0005)	(0.0017)	(0.025)
Exopalaemon styliferus	0.99	679.93	0.011	0.0022	0.0020	0.0014	0.0012	0.239
	(0.159)	(82.16)	(0.0033)	(0.0009)	(0.0010)	(0.0011)	(0.0010)	(0.080)
Varuna litterata	1.28	697.26	0.017	0.0023	0.0029	0.0005	0.0008	0.275
	(0.108)	(82.60)	(0.0097)	(0.0015)	(0.0010)	(0.0004)	(0.0009)	(0.033)
Polychaetes	1.49	1213.66	0.013	0.0016	0.0573	0.0066	0.0040	0.488
	(0.358)	(116.76)	(0.0018)	(0.0006)	(0.042)	(0.0018)	(0.0014)	(0.077)

Standard deviations are given in parentheses.

a predicted no-effect concentration (PNEC) for the target of concern (RQ = P(M)EC/PNEC), such that an RQ < 1 indicates a low, and thus acceptable risk, and an RQ ≥ 1 indicates a level of concern and possibly the deployment of management programmes.

Lam & Lam (2001) evaluated risks to faunal groups inhabiting the Mai Po mudflats and mangroves by comparing contaminant concentrations in the sediments with threshold effects levels (TELs) promulgated by the United States Environmental Protection Agency (USEPA, 1996). The TELs are defined as the concentrations below which toxic effects are rarely observed. These values were mainly derived from freshwater exposures of *Hyalella azteca* using 28-day survival, growth, and reproductive endpoints (USEPA, 1996). Where relevant data is available, RQs are calculated and tabulated in Table 4.

The risk assessment based on concentrations of various environmental contaminants in the sediments revealed that RQ values were all greater than one except for fluoranthene, chrysene and endrin in the mudflat; and benzo[a]pyrene and dibenzo(1,2,5,6)anthracene in the mangroves. RQ values for total PCBs were less than one. RQ values for copper and heptachlor epoxide were greater than 10. These results suggested that the levels of POC and certain metals in the Mai Po mudflats and mangroves might pose a substantial risk to aquatic organisms inhabiting the area. Results of this study further indicated that PCBs in the biota, except for *Uca arcuata*, posed a substantial risk to the waterbirds via food consumption (Lam & Lam, 2001). It was also observed that certain chlorinated pesticides (dieldrin and DDE, a metabolite of DDT) in the mudskippers (*Boleophthalmus pectinirostris*) were at levels that might cause harm to fish-eating birds (Lam & Lam, 2001). Tissue levels of PAHs were not directly relevant in this risk assessment process as many organisms, particularly fish, could rapidly metabolise and excrete PAH compounds.

In 2000, a second study, also commissioned by AFCD, was undertaken to examine the potential effects of waterborne pollutants on the breeding success of Ardeids in Hong Kong with special reference to the Mai Po/Inner Deep Bay areas (Lam et al., 2001). Possible exposure and effect pathways for bird populations in the Mai Po/Inner Deep Bay area are summarised in Fig. 3.

In this project, the feathers of two Ardeid species, the Little Egret (*Egretta garzetta*) and the Black-crowned Night Heron (*Nycticorax nycticorax*) were collected from six egretries and two egretries, respectively, located in different areas in the New Territories of Hong Kong, including the Mai Po Marshes. These feathers were digested and concentrations (μg/g dry weight) of copper (4.6–19.4), iron (8.1–641.3), manganese (0.4–19.4), zinc (51.3–183.5), lead (0.1–5.1), cadmium (0.01–0.15), chromium (0.06–1.7) and mercury (0.0–7.1) were determined by ICP-AES, ICP-MS and CVA-AS. A probabilistic risk assessment of the possible adverse effects on the breeding success of the Little Egret was

Table 4: Sediment threshold effects levels (TELs), maximum sediment concentrations, and calculated risk quotients (RQs) for the mudflat and mangroves in the Mai Po and Inner Deep Bay area.

	TEL[a]	**Maximum sediment concentrations on the mudflat (μg/g)**	**Maximum sediment concentration in the mangroves (μg/g)**	**RQ (mudflat)**	**RQ (mangrove)**
Trace elements					
Cadmium	0.6	1.59	0.7	2.65	1.17
Chromium	36.3	198.95	40.0	5.48	1.10
Copper	28	652.71	75.8	23.31	2.71
Lead	34.2	80.91	300.0	2.37	8.77
Manganese	615	No data	No data		
Mercury	0.17	No data	No data		
Zinc	94.2	233.96	553.3	2.48	5.87
PAHs					
Phenanthrene	0.042	0.057	0.079	1.36	1.88
Fluoranthene	0.111	0.073	0.239	0.66	2.15
Pyrene	0.053	0.107	0.320	2.02	6.04
Chrysene	0.057	0.016	0.116	0.28	2.04
Benzo(a)pyrene	0.032	0.107	0.029	3.34	0.91
Dibenzo(1,2,5,6) anthracene	0.032	0.040	0.025	1.25	0.78
Total PAHs	0.264	0.849	1.042	3.22	3.95
PCBs					
Total PCBs	0.032	0.028	0.018	0.88	0.56
Pesticides					
Chlordane	0.0045	0.0068	0.0065	1.51	1.44
Dieldrin	0.0029	0.011	0.0075	3.79	2.59
Heptachlor epoxide	0.0006	0.011	0.0062	18.33	10.33
DDE	0.0014	0.010	0.0043	7.14	3.07
Total DDT	0.007	0.00821	0.0081	1.17	1.16
Endrin	0.0027	<0.00001	0.0028	<0.0037	1.037

[a] From USEPA (1996); Table from Lam & Lam (2001).

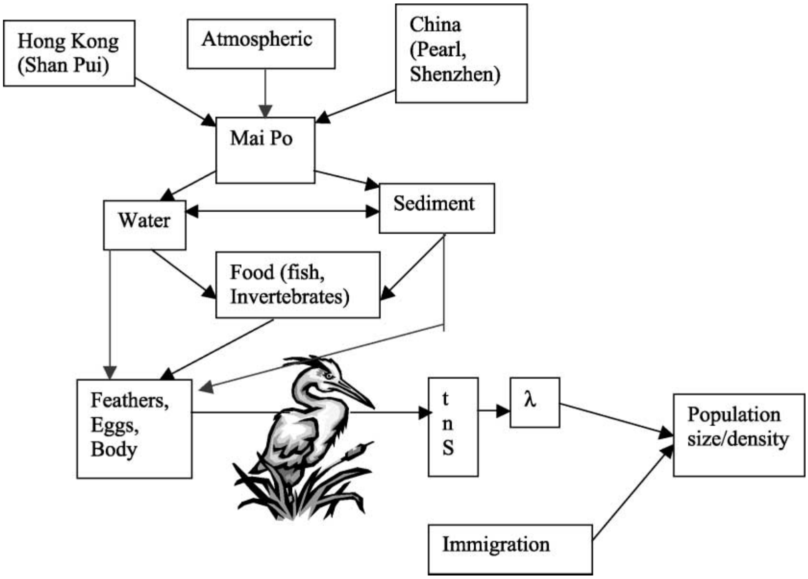

Figure 3: Possible exposure and effect pathways for bird populations in the Mai Po/Inner Deep Bay area. t is growth rate; n is fecundity; S is survivorship and λ is population growth rate.

carried out with respect to mercury, lead and cadmium. It was concluded that mercury (0.5–7.1 μg/g dry weight feathers) probably has had adverse effects at the Au Tau egretry of the Little Egrets, but no evidence of adverse effects at other egretries. Notwithstanding, the levels of lead and mercury were generally higher in the egretries close to the polluted Deep Bay. The probabilistic analysis also indicated a low likelihood of adverse effects of mercury on the breeding of the Black-crowned Night Herons at A Chau (0.3–1.2 μg/g) and Mai Po Village (0.0–1.4 μg/g). The evidence for the effects of lead and cadmium was limited but suggested there may possibly be adverse effects with lead but not cadmium. Details of this study are given in Connell et al. (2002).

In addition, POC, including PAHs, PHCs, PCBs and OC pesticides, were also analysed in bird eggs. Results of this study concluded that both species of Ardeid had concentrations of total DDTs present in the eggs sufficient to initiate adverse effects on the breeding success of these species (Connell et al., 2003). Some individuals in the populations were at higher risk due to the higher concentration of total DDTs present with maximum RQ of 9.0 for the Little Egret and 6.0 for the Black-crowned Night Heron. The Little Egret was at greater risk of adverse effects than the Black-crowned Night Heron since it had higher concentrations of the total DDTs present reflected in a higher RQ, i.e. 4.8 and 2.1 for the Little Egret and the Black-crowned Night Heron, respectively. The total PCBs present in eggs were at threshold levels where adverse effects could be initiated with the Little Egret. On the other hand, no effects would be expected with the Black-crowned Night Heron population. In addition, chlordane was at levels where adverse effects were possible, while the total hexachlorocyclohexanes (total HCHs) were at levels where no adverse effects on the breeding success of the Ardeids at Mai Po would be expected.

The above risk assessments are performed on a contaminant-by-contaminant basis. However, when target populations are exposed to a complex environmental mix (as is likely to be the situation for targets in the Mai Po/Inner Deep Bay area), it will be desirable to consider possible combined effects of contaminants and to take into account the possibility of interactions (additive, antagonistic, synergistic) in these combinations. If the total DDTs, total PCBs and mercury have additive effects the total effect on breeding success would be substantial for both the Little Egret and the Black-crowned Night Heron. Also, for complex mixes it may be possible that the most influential contaminant(s) has not been identified and hence not included in the risk assessment. This is an important area of research that deserves consideration for future investigation. It was clear from the results that the western waters were, in general, more affected by contamination of persistent toxic substances as compared to the eastern waters of Hong Kong (Lam et al., 2001).

8.5. Recommendations

Based on the risk assessment results available to date, a number of recommendations can be made to further elucidate the fate and environmental effects of contaminants in the Mai Po and Inner Deep Bay area.

(1) Regular monitoring of mercury and DDTs in the Ardeid tissues and in key environmental compartments (e.g. water and sediments) of the Mai Po and Inner Deep Bay system should be undertaken by relevant authorities to provide an early warning of potential adverse effects due to increase in concentrations of these contaminants.
(2) Since there is some evidence that Ardeids inhabiting the eastern and western parts of Hong Kong may be exposed to different levels of contamination, a strategic monitoring of concentrations of specific toxicants in Ardeids inhabiting the eastern and western waters will provide additional information on the potential effect of water pollution on the Ardeids.
(3) Although there is no clear evidence to date to suggest that the breeding success of Ardeids in Hong Kong is impaired by environmental contaminants, the breeding success of Ardeids from major nesting sites in Hong Kong should be

regularly monitored. In the event that a significant decrease in the breeding success of a specific Ardeid population is associated with an increase in the concentrations of the key contaminants (e.g. DDTs and Hg), an inventory of the usage and occurrence of these compounds in the Deep Bay and Mai Po area should be developed. This is particularly important given that there is no reliable information on the source of specific contaminants in Hong Kong.

(4) As a precautionary measure, levels of cadmium, chlordane and lead should also be monitored regularly, and the possible effects of these chemicals on the breeding success of waterbirds carefully evaluated.

(5) Policies, methods and procedures for the management of important toxicants in the Mai Po system should be developed. Information collected from above should be synthesised and used to recommend practical measures to reduce the impacts of toxic pollutant residues on the Ardeids in the event that a significant effect due to environmental contaminant is clearly demonstrated. The aim here is to formulate a management plan that is cost-effective and practical. To this end, relevant government officials, non-government organisations, green groups, and other stakeholders should be widely consulted.

8.6. Overall Conclusion

Environmental pollution has long been considered a major threat to the long-term sustainability of our coastal environment, including areas of high ecological importance and conservation value. Specifically, it is apparent that PAHs, OCs as well as certain trace metals may pose a risk not only to the marine ecosystem, but also primary and secondary consumers of marine organisms. Although there are now considerable data on the levels of common environmental contaminants in the marine sediments and, to a lesser extent, other environmental compartments, such as biota, there is still a general paucity of information on the sources of these chemicals and their precise effects on biological systems in Hong Kong waters.

In regard to sources of environmental contaminants in Hong Kong, there is evidence that the marine sediments around the Mai Po/Inner Deep Bay area are contaminated by high levels of metals and certain persistent organic pollutants, including banned compounds, such as DDT. The sources of these contaminants are still not clearly known. Particularly, the importance of atmospheric input to the western waters of Hong Kong from the industrial areas in the Pearl River Estuary will need to be examined. Further investigations should focus on identifying the sources of important environmental contaminants in Hong Kong waters, and elucidating the effects of these chemicals on local biological/ecological systems.

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Chapter 9

Modelling Contamination in an Urban Canal Sediment: Some Preliminary Results from a Phytoremediation Project

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Abstract. Sediments in a derelict section of canal in a former industrial region of North-West England contain a wide range of elevated contaminants including Cu, Zn, Ni, As, Pb, Cd, Cr, mineral oils, TPHs, PAHs and sulphides. Associated costs of disposal have been a constraint to restoration of the canal, which has remained unused for navigation for 50 years or more. On-site phytoremediation is being used in the current project to investigate whether a healthy environment can be restored without extensive removal of the sediment from the site. A raised platform of dredged sediment has been created within the partially drained canal. As the sediment dries and becomes aerated, metal availability was markedly altered and volatilisation rates of organics appeared to increase. Decreasing sulphide/sulphate ratios, lowered pH and altered Fe mobility had differing effects on trace elements. Repeated wetting and drying mobilised a substantial proportion. The project is comprehensively modelling these processes, and aims to demonstrate that metals can be rendered immobile and non-hazardous in soils and biomass whilst plant roots and developing biota optimise conditions for the natural attenuation of organics.

9.1. Introduction

Canals were formerly important transport routes for industry and trade in North-West England. Between Liverpool and Manchester, canals were excavated from

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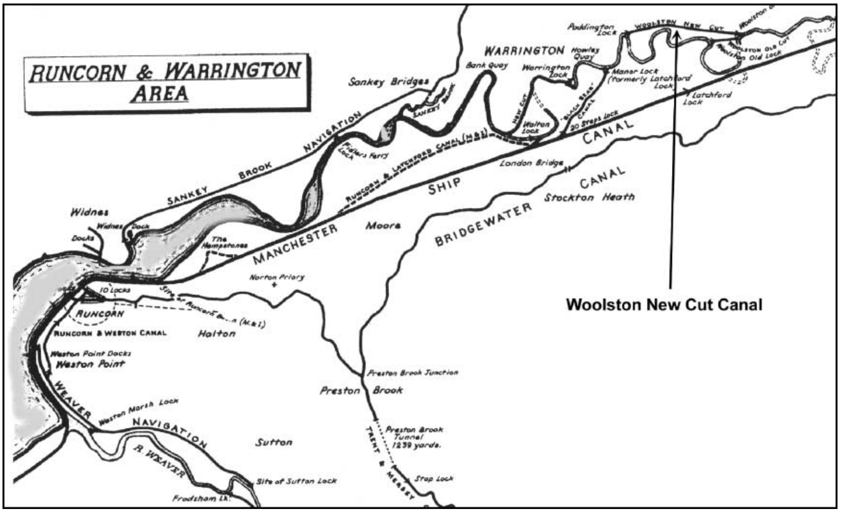

Figure 1: Old map showing extensive canals in the region, including the 2 km section of the Woolston New Cut Canal excavated in 1821 to improve navigation of the River Mersey. Liverpool and the open sea are to the left of the map.

the 18th Century to improve navigation of the River Mersey which links the two cities with each other and to the North Atlantic Ocean (Fig. 1).

The Manchester Ship Canal was excavated in the late 19th Century as difficulties with navigation of the tidal river were losing trade to the new railways. The 2 km section of canal at Woolston, the site of the present study, was excavated in 1821 to shorten large bends in the River Mersey and to improve navigation. The river at Woolston is still subject to tidal influence; water into the canal was formerly controlled by lock gates and by an aqueduct which crossed the river. Industrial development along the Woolston New Cut Canal appears to have been limited to chemical works, a gunpowder factory, a tannery and an abattoir, although adjoining sections of the riverbank have supported various industrial and engineering works, more tanneries, and a gas works. The sediment was undoubtedly influenced by numerous industries and by spillages from ships. Perhaps associated with the demolition of the aqueduct in 1978, the canal has fallen into disrepair and dereliction. The canal ceased to flow, water levels dropped and the standing water and wet sediment has been colonized by vegetation: dominantly *Typha latifolia* (Reedmace), with *Salix atrocinerea* (Sallow) at the edges.

After abandonment, environmental improvement of this derelict section of canal was prohibitively expensive due to the cost of disposal of some 40,000 t of contaminated sediment. From an earlier unpublished consultancy report, it was known that the canal sediment consisted of a wet, black, odorous and oily mud (up to 1.5–1.7 m depth) containing a wide range of elevated contaminants including Cu, Zn, Ni, As, Pb, Cd, Cr, phenols, mineral oils and S. The proportion of bioavailable metals in this type of sediment may be as much as 40% of total concentrations, even after 60 years without disturbance (Stephens et al., 2001). This may present a considerable hazard of dispersal to the wider environment if wet, reduced sediments are disturbed. The context of this current work is the restoration of the canal and adjacent land to community use for recreation and amenity, whilst addressing residual contamination issues.

Phytoremediation is receiving considerable attention as a low-cost treatment technology for land and groundwater contaminated with heavy metals (Chaney et al., 1997; Glass, 1999; Vangronsveld et al., 2000; Pulford et al., 2002) and organic compounds (Carman et al., 1998; Meagher, 2000; Campanella et al., 2002; Susarla et al., 2002). One strategy is to use fast-growing trees, particularly *Salix* and *Populus*, to remove, stabilize or enhance the volatilization of polluting chemicals (Jones et al., 1999; Greger & Landberg, 2000; Pulford et al., 2002; Vervaeke et al., 2003). There is a real possibility that plants can be used to reclaim contaminated land and restore sustainable and healthy soils (Kearney & Herbert, 1999; Dickinson, 2000). The project described in this chapter is a case study of the feasibility of using phytoremediation as a low-cost alternative to cart and dump of the contaminated sediment. By modelling this ecosystem, the objectives are to investigate whether metals can be rendered immobile and non-hazardous in soils and biomass, whilst plant roots and developing biota optimise conditions for the natural attenuation of organics.

9.2. Methods

Established vegetation was cleared from a 150 m section of the New Cut Canal bank and from shallow sediment on the side of the canal opposite the towpath. A raised platform (3.5 m wide) above the existing water level was then created along the shallow side, by dredging and transfer of sediment from the towpath side to the shallow side of the canal (Fig. 2). The platform was divided into six experimental blocks. Within each block, 12 short-rotation coppice taxa (species, hybrids or clones) of willows, poplars and alders (Table 1) were each planted in double rows (0.5 m apart), randomly selected, with 1 m between each double row. An additional unplanted space, equivalent to a double row, was left unplanted within each block, as a control. In each row, there were six plants of each taxon, planted 0.5 m apart. *Salix* and *Populus* were planted as pegs, and *Alnus* was planted as 50–70 cm rooted stock (pruned back after establishment).

Three sediment samples (0–15 cm) were taken with an auger between each row, and then bulked for each of the six blocks. The samples were thoroughly

Figure 2: Canal being dredged to create a 3.5 m raised platform within the canal, with double rows of planted trees.

mixed, then evenly divided and delivered to three national UKAS accredited laboratories. A more extensive modified herringbone design was used to sample sediments in order to map spatial variation of metals on the planting platform. The sediment samples were all taken between trees within the planted rows, from four set distances across the width of the planting platform. This provided one sample from within each planted row of six trees, using a modified herringbone design along the length of the platform. Subsequent chemical extractions (*aqua regia*, EDTA and $CaCl_2$) and metal determinations were carried out in-house, with some support from the NERC ICP facility at Royal

Table 1: Tree species and clones planted on the raised platform of sediment in the canal.

Salix viminalis 'Jornn'*
Salix viminalis × *schwerinii* 'Tora'*
Salix caprea × *cinerea* × *viminalis* 'Calodendron'*
Salix viminalis × *burjatica* 'Ashton Stott'*
Salix viminalis × *caprea* 'Sericans'
*Salix fragilis**
Salix atrocinerea[a]
Populus deltoides × *nigra* 'Ghoy'
Populus trichocarpa 'Trichobel'*
Alnus glutinosa
Allnus incana
Alnus cordata

*Recommended for SRC use in FC Inf. Note 17.
[a] From cuttings of trees naturally colonizing canal.

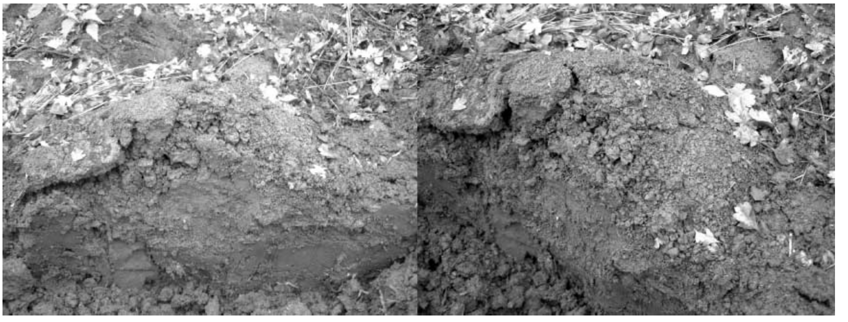

Figure 3: Vertical profiles through surface "puffs" of friable soil, surrounded by more compacted soil on the planting platform. Tree leaves (approx 3 cm length) provide scale.

Holloway. Independent analyses were also carried out using AAS in the Chemistry Department at the University of Glasgow.

After the first year, physical changes in the sediment were apparent. Some areas were flattened through compaction (at least partly through trampling on the platform), but some areas had small upwellings of loosely compacted sediment-referred to here as puffs (Fig. 3). Samples of these two types of sediment were also compared with freshly collected benthic sediment. In this case, metals were extracted using HNO_3/HCl (9:3), 0.05 M EDTA or deionised water, followed by AAS determination of Fe, Zn, Cu, Cd and As (the latter using hydride generation). Additionally, freshly collected air-dried (35°C for 24 h in an air-circulating oven) sediment was re-wetted (5 g sediment in 25 ml deionised water), continuously agitated for 1 hour, and then the filtered leachate was analysed for the same elements. The sediment was then air-dried and the process was repeated eight times. All treatments and analyses were carried out in triplicate. Only selected results are shown in the present chapter.

Five boreholes were established to the side of the canal to monitor groundwater, for analysis by an external laboratory. Establishment of growth of the trees was monitored during the first year, and invasive plants were controlled by hand weeding and herbicide spraying.

9.3. Results and Discussion

The first analyses of the sediments showed considerable variation between laboratories (Table 2). Part of the explanation for these differences concern different analytical methodologies, the details of which were not automatically provided by any of the laboratories. Whilst the three laboratories were UKAS

Table 2: Mean values (mg kg^{-1}) for sediment contamination as provided by separate UKAS accredited laboratories. Values in bold are those exceeding standard contamination thresholds ($n = 7$).

Determinand	**Laboratory**		
	A	**B**	**C**
Sulphate	0.53	0.60	**0.97**
Sulphide	**1078**	354	84.0
Arsenic	**349**	**682**[a]	**1.6**
Boron	**69.0**[a]	**3.5**	1.6
Cadmium	**12.7**	**13.6**	**18.1**[a]
Chromium	**790**[a]	**1471**[a]	**977**[a]
Copper	**567**[a]	**1076**[a]	**736**[a]
Lead	**1221**[a]	**2150**[a]	**1443**[a]
Nickel	**67**	**78**	**783**
Mercury	**3.8**	**7.5**[a]	**4.8**
Zinc	**3631**	**5835**[a]	**4286**
Cyanide	103[a]	10.6	23.2
Total PAH	**216**[a]	**141**	**121**
TPH	**7636**[a]	**5671**[a]	2207[a]

[a] Significantly different to results of other laboratories.

accredited, standard methods for sample preparation, extraction and analysis vary (Dickinson et al., 2000). Although there is a general consensus as to which determinands exceed existing guidelines and thresholds (such as the recently superceded UK ICRCL guidelines), significant differences existed between the laboratories for every determinand. The only exception was for sulphide where data were very variable between samples; this variability probably masked differences between laboratories. More detailed sediment sampling and analysis in-house showed that mobile pools of metals varied considerably (Table 3).

A particularly large proportion of the phytotoxic metal Zn appeared to be in a bioavailable form. Most Fe was strongly bound in the sediments (Fig. 4), but a decrease in the more mobile fractions was evident in the surface of the sediment as it dried. This change was reflected in an increased amount of oxide-bound Fe (data not shown). The changing chemical conditions reduced the mobility of Cu, probably as it became bound by chemisorption. In contrast, Zn is physisorbed and this metal became more mobile; exchangeable and pore water concentrations of Zn doubled and this metal moved from the surface layers as the sediment dried and oxidized.

Table 3: Range of metal concentration across the planting platform and relative bioavailability in sediment (pH 3.6–5.5).

Metal	Total concentration (mg kg^{-1})	EDTA-extractable as % of total concentration
Cu	418–1,003	0.6–9.6
Pb	776–1,555	1.1–8.9
Zn	1,003–6,565	11.8–46.5
Ni	55–130	5.0–27.4
Cd	6.9–23.8	1.7–20.9
Cr	496–1,081	<1

The preliminary results suggest that some metals, such as Zn, in this canal sediment may cause toxicity to plant growth. Whilst there were visual symptoms of toxicity in the foliage of some plants, establishment and survival were good. Mortality was 22% in the first year, marginally better than is normally expected for short-rotation coppice on clean soils. Interestingly, the least successful species was *S. atrocinerea* (67% mortality); this species had naturally colonized the canal banks and exposed sediment, but appeared to be a difficult species to root from cuttings. Plots containing this species were not replanted, but instead are being maintained free of weeding and herbicide sprays as a "natural regeneration" treatment.

Sediment pH of the highly organic sediment on the planting platform was markedly lower than that of freshly dredged sediment (Table 4). Aeration and drying of the sediment substantially increased sulphate, with increased mobility of metals in sediment from the planting platform, but reduced mobility of As.

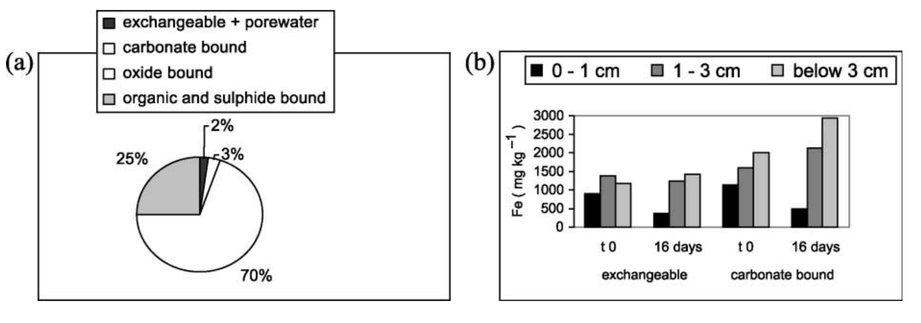

Figure 4: (a) Fe fractionation in freshly dredged wet sediment, and (b) change in the two more labile fractions after 16 days' exposure to air.

Table 4: Analysis of freshly dredged sediment and the two types of sediment on the planting platform.

	pH	Org. matter (LOI %)	Water extracts (mg kg^{-1})					
			SO_4	Fe	Cu	Zn	Cd	As
Benthic sediment	5.7	33.1	83.6	7.18	0.20	2.94	nd	2.50
Planting platform puffs (friable sediment)	3.3	28.6	9,405	114	55.3	29.1	2.71	0.24
Planting platform (compacted sediment)	3.5	31.5	1,190	172	13.5	29.3	1.14	0.78

nd = not detectable.

However, continued wetting and drying of the sediment probably better reflects conditions on the site. The amount of metals in the leachate declined with repeated wetting and drying, as the more labile pools become depleted, Zn became more mobile by about the fifth cycle (Fig. 5).

Using this simple and rather crude method of extraction, a substantial proportion of the total amount of each element was removed from the sediment samples (Table 5). Further work is required to understand how these elements move deeper into the soil profile, and perhaps back to the benthic sediment in the open canal. In the second year the open water had been entirely colonized by *Typha*, where metals would be expected to become immobilized around the roots (Ellis et al., 1994; Ye et al., 1997).

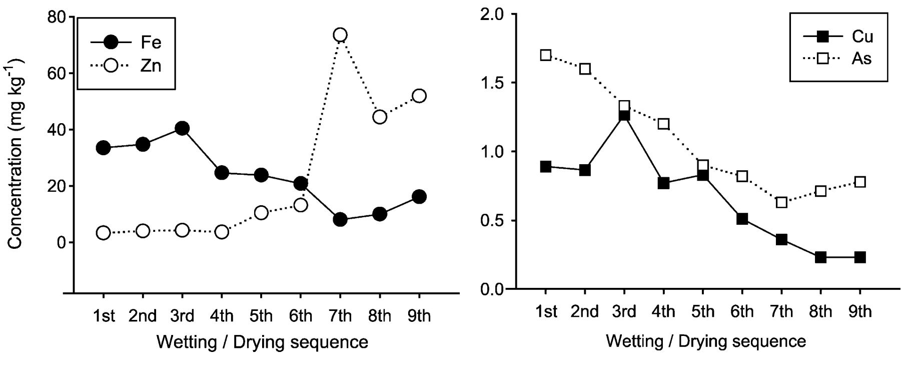

Figure 5: Concentration of elements in leachate after repeated air-drying of freshly-dredged sediment.

Table 5: Element removal from 5 g sediment sample following nine repeated wettings and dryings of freshly-dredged sediment with deionised water.

	Total concentration ($\mu g\ g^{-1}$) in original sample	µg Removed	Percentage of total removed (%)
As	222.3	241.8	22
Cu	96.7	148.3	31
Zn	3899	5216	27
Fe	6661	5303	16

9.4. Conclusions

Since the canal became derelict, the contaminants in the sediment have been maintained in an anoxic, reducing environment. After dredging and exposure to air, the new aerobic and oxidizing environment will undoubtedly induce a host of chemical and biological changes. Sediments deposited on land following dredging have been the subject of previous studies in Belgium (Tack et al., 1996; Tack et al., 1998, 1999; Tack & Verloo, 1999) and the UK (Stephens et al., 2001). Most metals show redistribution from residual to mobile phases during drying and oxidation that is also associated with decreasing sulphide/sulphate ratio. Fe decreases in the surface layers as the sediment dries and becomes aerated and acidified. Mobility of metals clearly differs, but simple experiments showed that 16–31% of elements were fairly rapidly mobilised in water after repeated wetting and drying for 10 days. However, this modelling is in the early stages and more experimental work is required to establish long-term prediction of metal migration.

In the present project, the sediment was retained within the canal banks, and it is likely that migration of metals will be controlled to a large extent by the clay liner of the canal. No elevated metals or organics were recorded in the borehole water after the first year. Nevertheless, before this can become a treatment technology, it is important to demonstrate that contaminants are not quickly dispersed to the wider environment. Another potential source of dispersion is through food chains (Vandecasteele et al., 2002, 2003). Manipulating the processes of contaminant dispersion or immobilisation offers a real possibility of treating contaminated sediment without removal, whilst contributing to a healthy, sustainable, non-hazardous landscape of high ecological and amenity value. One possibility is that sediment disturbance in this way may enhance volatilization of organics whilst actually concentrating metals in a much reduced benthic sediment, within the *Typha* rhizosphere. In turn and if necessary, this lesser amount of sediment and its associated vegetation could be removed at a much reduced cost, compared to disposal of the currently existing sediment. This may be a step towards a realistic, generic and transferable methodology with wide application for cost-effective reclamation of contaminated sediments.

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Session III

Wetland Management Strategies in Asia

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Chapter 10

Conflicts in the Management of a Wetland Nature Reserve — Case Study of the Mai Po Nature Reserve, Hong Kong

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Abstract. Objectives are set in the management of any nature reserve and at times, apparent conflicts may arise where the achievement of one objective may impact on another objective. For example, whilst mudflats and coastal mangroves are important conservation habitats, the spread of mangroves over the mudflats may have to be controlled in order to maintain the area of the two habitats in balance. The reasoning behind such apparent conflicts have to be carefully explained to visitors otherwise, they will depart with a negative impression of the reserve. Therefore, education and public awareness about the management of the reserve is just as important as maintaining the ecological and cultural values of the site itself. This chapter uses the Mai Po Nature Reserve (MPNR), Hong Kong SAR to describes some of the apparent conflicts that may arise in the management of a wetland nature reserve.

10.1. Introduction

Nature reserves around the world are subject to some form of management intervention. This ranges from local people who farm, graze their livestock, or harvest resources from the site as part of their livelihood, to government representatives or private individuals who follow a prescribed plan to maintain the site's ecological and cultural significance. In the case of the latter, the reserve staff manage the site in order to meet a number of goals which could, unless the management is done carefully, cause apparent conflicts with each other.

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Management of the MPNR, Hong Kong is used as an example to illustrate the types of apparent management conflicts that may occur in a wetland nature reserve, and the steps that can be taken to reduce those conflicts.

10.1.1. Geography

The Inner Deep Bay wetlands are located at the eastern edge of the estuary of the Pearl River, southern China. The Bay is bounded to the north by the Shenzhen Special Economic Zone (SEZ) and to the south, by the Hong Kong Special Administrative Region (SAR). During low tide, 2,700 ha of mudflats are exposed in the Bay, fringed by an area of some 400 ha of inter-tidal mangrove forest. Behind the mangrove, are traditionally operated shrimp ponds (locally called *gei wai*, which support stands of mangroves and reedbeds), and commercial fishponds. Since the mid-1970s, the area of these wetlands have declined due to their gradual in-filling for urban developments.

Protection of these wetlands on the Hong Kong SAR side of Deep Bay began in 1976 when the *gei wai* and mangroves at Mai Po were designated as a Site of Special Scientific Interest (SSSI). Management of this site for conservation and for promoting environmental education began in 1984 when WWF Hong Kong began to take over management of the *gei wai*. Protection was further increased in 1995 when a 1,500 ha area of the wetlands (including the MPNR), was designated a Wetland of International Importance under the Ramsar Convention (Fig. 1). On the Shenzhen SEZ side of the Bay, the Futian National Mangrove Nature Reserve was established in 1984.

10.1.2. Ecological Importance

The wetlands of Inner Deep Bay are best known as a wintering site for up to 68,000 waterbirds, and another 20,000–30,000 shorebirds which use the site as a staging post during spring and autumn migration. In addition, 18 species of these waterbirds are considered threatened and 30 species occur in numbers that are greater than 1% of their estimated population in East Asia (Carey and Young, 2001).

However, the Inner Deep Bay wetlands are also important for supporting other wetland wildlife, such as the Eurasian Otter *Lutra lutra*, as well as a number of rare and endangered invertebrates, e.g. the endangered dragonfly *Mortonagrion hirosei* (Young, 1999).

Apart from wildlife, the wetland habitats at Mai Po and surrounding areas are also important (Table 1).

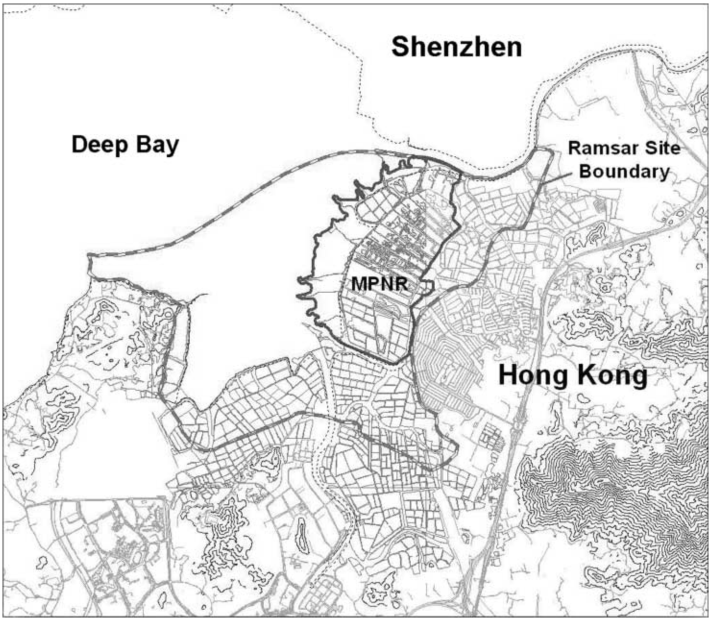

Figure 1: Map showing the Mai Po Inner Deep Bay Ramsar Site (Mai Po Nature Reserve, MPNR).

10.2. Management of the Inner Deep Bay Wetlands

10.2.1. Historical Management

The local people living around the coast of Inner Deep Bay have been managing the area's wetlands since at least the 1200s, when the first settlers established themselves at San Tin. These people mainly depended on the local fisheries for their livelihood, such as fish, shrimps, crabs and oysters.

From the early 1900s, the landscape began to change, with groups of immigrants from mainland China coming down to settle along the southern (Hong Kong) coast of Deep Bay at various times. Each time such people came, they brought with them new techniques of reclaiming and farming the land, such as:

- 1920s — techniques for brackish rice farming;
- 1940s — shrimp farming using tidal ponds called *gei wai*;
- 1960s — pond-fish farming.

Table 1: Importance of the habitats at Mai Po Nature Reserve and surrounding areas.

Habitat	**Approximate area (ha)**	**Importance**
Deep Bay mudflats	2,700	The most important feeding habitat for the migratory waterbirds that visit Deep Bay
Inter-tidal mangroves	400	The largest mangrove stand in Hong Kong and the sixth largest protected stand in China.
Gei wai shrimp ponds	240	One of the last remaining areas of traditionally operated shrimp ponds not only in South China, but also in Asia.
Reedbeds	45	The largest area of reeds in Hong Kong, and probably one of the largest stands in Guangdong Province.
Commercial fishponds	1,200	Traditionally operated fishponds are as an example of the wise use of wetlands and are of high ecological value.

As a result, the Deep Bay wetlands went through a series of land-use changes, and the remains of each type of these farming practices can still be seen today in the landscape (Irving and Morton, 1988).

10.2.2. Management Plan for the Mai Po Nature Reserve

The Mai Po Marshes have been well know as a place for migratory waterbirds since the late 19th century. Steps to look into the designation of the Marshes as a protected area began in the 1960s but in 1975, the colonial Hong Kong Government approved a large-scale housing development adjacent to the Marshes. This raised the concerns of the conservationists at the time over the impacts from disturbance if residents from the development had unrestricted access to the Marshes. After lobbying from these conservationists, the Government declared the Marshes as a restricted access area the same year. In 1976, the Hong Kong Government further designated the Mai Po Marshes as an SSSI. However, this was only an administrative designation and did not confer any real protection to the site, nor was the site managed for conservation.

The Mai Po Marshes are made up by inter-tidal shrimp ponds (*gei wai*), created in the early 1940s. However, by the late 1970s, they were becoming increasingly unprofitable due to competition from pond-fish farming and the gradual pollution of Deep Bay due to the catchment becoming more urbanised. As a result in 1984, WWF Hong Kong began to take over the management of these *gei wai* for conservation and for promoting environmental education. A special committee within WWF Hong Kong was formed to oversee the development and management of the MPNR, with members from various government departments, academics, green groups and interested persons. The goals for the Reserve as outlined in the management plan (Young, 1999) are:

1. To manage the MPNR so as to maintain and, if possible, increase the diversity of habitats appropriate for south China lowland wetlands, and the richness of native wildlife in the area.
2. To promote the use of the area for educational purposes both by students and the general public (including the provision of special facilities and tours for the disabled).
3. To realise the training potential of the Reserve as part of the Ramsar Site so as to promote wetland conservation and wise use in the East Asia/Australasian Flyway, in particular China.
4. To promote scientific research relevant to the management and conservation of wetlands and their biota.
5. To promote, and support measures to reduce and minimise external threats to the habitats and wildlife at the Reserve.

In order to achieve Goal 1, a number of Management Objectives were set out in the management plan:

1.1 To provide suitable roosting and feeding habitats for Black-faced Spoonbills.
1.2 To provide suitable high tide roosting sites for a significant population of the shorebirds in Deep Bay.
1.3 To provide suitable roosting and feeding sites for a significant population of the wintering waterfowl in Deep Bay.
1.4 To ensure suitable habitats for key species.
1.5 To maintain and manage the mangrove habitats.
1.6 To maintain and manage the reedbed habitats.
1.7 To develop a series of freshwater habitats within the reserve.
1.8 To maintain the traditional operation and landscape of the *gei wai* habitats.
1.9 To maximise biodiversity without compromising the above objectives.
1.10 To monitor the progress of all habitat management on the reserve.
1.11 To encourage research projects that will achieve the above objectives.
1.12 To review regularly the management plan in the light of results from the monitoring programme, research and changing circumstances.
1.13 To abide by local legislations and meet obligations under agreed international conventions and relevant inter-governmental agreements.

In 1995, the Hong Kong Government designated a 1,500 ha area of wetlands around Deep Bay as a Wetland of International Importance under the Ramsar Convention (Convention on Wetlands), and the MPNR was incorporated as part of the Ramsar Site.

A year later in 1996, the Hong Kong Government began providing annual subvention to WWF Hong Kong for the wetland habitat management they were carrying out at Mai Po. With the completion of a management plan for the Mai Po Inner Deep Bay Ramsar Site in 1997, WWF Hong Kong had to ensure that the work programme laid out in the Mai Po Management Plan complemented that in the management plan for the larger Ramsar Site. With the formation of a Wetlands Advisory Committee (WAC, with a separate Scientific sub-committee and a Management sub-committee) under the Agriculture, Fisheries and Conservation Department (AFCD) in 1998, it meant that the habitat management work at the Reserve would be scrutinised by both the WWF Hong Kong Mai Po Management Committee as well as the WAC Management sub-committee.

10.2.3. Management Plan for the Ramsar Site

The 1997 management plan for the Mai Po Inner Deep Bay Ramsar Site (Anon., 1997) divided the Site into five zones:

Core Area to provide an undisturbed, largely natural reference area. Maintenance of natural processes has priority and access is generally limited to essential management, monitoring and research purposes.

Biodiversity Management Zone To provide a refuge for waterfowl (including a high tide roost) and a focus for biodiversity conservation, education and training in a relatively intensively managed environment.

Wise Use Zone To allow ecologically sustainable use of wetland and other natural resources to be carried out in a way compatible with the Ramsar Site management goals and objectives and, where appropriate, to be encouraged and promoted.

Public Access Zone To enable people to have unrestricted (but managed) access to a part of the Ramsar Site in order to appreciate its special values and enjoy contact with wildlife.

Private Land Zones To recognise the existing legal status of the land.

Each of the above zones is divided into smaller compartments, including the "Biodiversity Management Zone" which includes the MPNR. At Mai Po, each of the compartments consists of a group of *gei wai*, each with its own broad management intention but without any detailed management prescription (Fig. 2; Table 2).

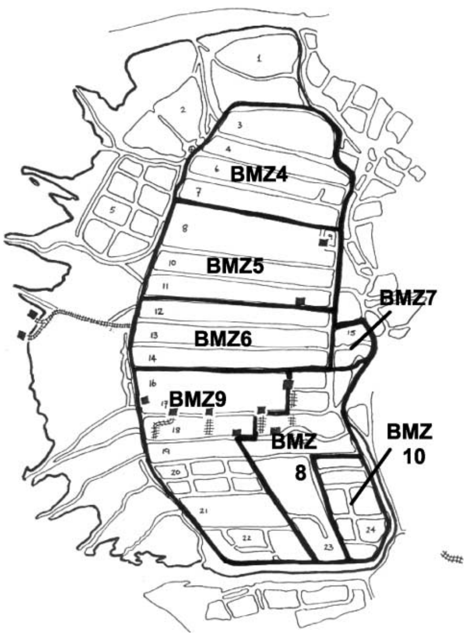

Figure 2: Management compartments within Mai Po Nature Reserve.

10.3. WWF Hong Kong Management of Mai Po

10.3.1. Vegetation Management

Although the plant communities at Mai Po are ecologically important in their own right (i.e. mangroves and reedbeds), they also provide shelter, nesting and feeding sites for wildlife. However, these plants need to be managed and their spread controlled whenever they begin to encroach into other habitats of importance. Examples include:

- reeds encroaching into open areas of water within the *gei wai* that are used by waterbirds,
- reeds into designated mangrove habitats,
- mangroves into designated reedbed habitat, and
- mangroves over the mudflat which is the most important habitat for waterbirds in the Ramsar Site.

Apart from the above examples, other types of vegetation may have to be removed because they are exotic or invasive species, and their spread will also reduce the diversity of habitats and species in the wetland. Such species include

Table 2: Management intentions of the compartments of the Biodiversity Management Zone at Mai Po Nature Reserve.

Compartment	*Gei wai* #	Management intention
BMZ 4	3,4,6,7	In the medium term, to adjust conditions in favour of supporting higher numbers of Black-faced Spoonbills.
BMZ 5	8,9,10,11	In the medium term, to adjust conditions in favour of supporting a substantial block of reedbed habitat (plus the small existing patch of bulrush vegetation).
BMZ 6	12,13,14	In the long term, to maintain traditionally managed production *gei wai* with areas of mangrove vegetation.
BMZ 7	15a, b	The long-term intention is to maintain and improve the Education Centre and its associated wildfowl collection.
BMZ 8	15c, 16/17 (east), 18 and 19 (east), 23	In the medium term, to adjust conditions in favour of creating an open freshwater area.
BMZ 9	16/17 (west), 18 and 19 (west), 20, 21, 22	In the medium term, to adjust conditions in favour of creating an open, tide-influenced area, whose primary objective is to provide a secure high tide roosting area for waterbirds.
BMZ 10	24	In the medium term, to adjust conditions in favour of creating a series of freshwater lakes of varying depth with surrounding areas of marsh.

the exotic invasive climber *Mikania micrantha*, the grass *Spartina* spp., and the mangrove *Sonneratia* spp.

Vegetation Control. Vegetation control has been carried out using one or more of the following techniques:

- Physical clearing with or without machinery.
- Controlled burning. This can be used to prevent colonization by certain plants (e.g. heat sensitive trees and shrubs) and to rejuvenate grasslands and reedbeds. Fire is a powerful management tool but needs to be used carefully. This is not only for safety reasons, but also for public relations reasons as many people have a negative impression of fires in the countryside and would initially consider that fire should not be used as a management tool within nature reserves.

- Herbicide. The herbicide chosen must be carefully selected to have minimal impact on the environment. Again, many members of the public have a negative impression over the use of chemicals in a nature reserve so care must be taken used in their use. Ideally, the reserve should carry out their own tests on the toxicity of the chemicals, and be able to discuss the benefits and potential harm that they may cause.
- Controlled grazing by domestic stock or control harvesting of plant products can sometimes be used to maintain desired vegetation such as grassland. If closely controlled, they can also be used to produce a uniform stand of vegetation that can provide feeding or nesting sites for wildlife. However, there may be a problem of the spread of diseases and parasites between domestic stock and wildlife. The stock should therefore be carefully monitored to ensure that the protected area does not lose any of its original values due to their presence.

Tree Management. Prior to the early 1980s, the landscape within the present MPNR used to be open, with very few trees growing along the *gei wai* bunds. This was because the local fishermen regularly used the bunds as footpaths to access the sluice gates at the seaward end of the pond where they had their homes and where they would harvest *gei wai* shrimps. Each winter, the fishermen would cut and/or burn the vegetation along the bunds in order to stop them from being overgrown.

With the completion of the Frontier Closed Area (FCA) Border Fence Road at Mai Po around 1982, fishermen stopped using the bunds as footpaths and instead, would drive along the new road to access their shrimp ponds. As a result, trees began to grow up along these bunds so that today we have a well-vegetated landscape. However, many of the waterbirds for which Mai Po is important (e.g. ducks and shorebirds), prefer to use open areas with few tall trees, in order to more easily detect predators. One example of the benefits of tree management is from *gei wai* #16/17 which between the late 1980s and early 1990s, was used by up to 10,000 migratory shorebirds in Spring as a high tide roosting site. From 1994 however, a decreasing number of shorebirds began to use this pond, and by 1997 shorebirds completely abandoned the pond. The main reason for this problem was suspected to have been because of tall trees having grown up along the bunds of the pond. When these trees were removed in early spring 1997/98, shorebirds began using the pond again.

As a result, WWF Hong Kong has a programme to manage the tall trees along the sides of the Mai Po *gei wai*, especially those ponds that are managed as waterbird habitat. However, there will likely be negative public opinion about the removal of trees from within a nature reserve. Therefore, this programme needs to be carried out carefully, by removing any tall trees in small blocks or selectively along different *gei wai* bunds, rather than removing a long line of tall trees along a bund in one single operation. This is so as to minimise any sudden, negative visual impact to visitors from the removal of large blocks of trees from the reserve.

Tree Planting. Whilst there is a programme of tree removal from the bunds at Mai Po, there is also a programme of tree planting next to footpaths and around the landward edge of the reserve. This is mainly so that the trees can provide shade for visitors and act as a visual screen against nearby developments. In selecting the tree species for planting, priority will be given to species which are native, are associated with lowland wetlands, and can provide fruit for frugivorous birds. These have included species such as *Ficus superba* and *Sapium sebiferum*.

However, as discussed in "Tree Management" it is considered that there is now sufficient tree cover within Mai Po, and so shrubs are planted instead of trees. Examples of shrubs include *Rhaphiolepis indica* and *Schefflera octophylla*, which are also fruit bearing species.

Invasive Climbers. There are a number of invasive climbers at Mai Po which if left unchecked, will smother and kill the mangroves and other plants at the reserve. These species include *Derris alborubra*, *Ipomoea* spp., *Mikania micrantha*, *Paederia scandens*, *Passiflora foetida* and *Strophanthus divaricatus*.

Trials have been undertaken to identify the most effective way of controlling the most harmful of these species, *Mikania micrantha*. These trials involve clearing the *Mikania* by mechanical cutting, spraying with herbicide, burning and a combination of these techniques in autumn. So far, the results indicate that removal is most effective by using a combination of spraying and then burning.

Reedbed Management. The 46 ha of reed grass *Phragmites australis* at Mai Po is the largest area of this habitat remaining in Hong Kong, and probably the largest area in Guangdong Province. Over the years, the area of reedbeds at Mai Po has increased as the *gei wai* have silted up.

Apart from being an important ecological habitat that supports a diversity of wildlife, reeds also have a commercial value. In many parts of Mainland China, reeds are harvested for thatching the roof of houses, paper production and for making traditional Chinese herbal medicine. Harvesting and removal of the reeds also has the benefit of reducing leaf litter build up, and thus the rate of siltation.

A number of strategies have been successfully developed by nature reserves around the world for managing reedbeds for wildlife. This includes managing the water levels in the reedbed, or by managing the reed itself by a mixture of spraying, cutting and burning at different times of year, on rotation varying from 1–15 years (Fry and Londale, 1991; Kirby, 1992; Burgess et al., 1995; Hawke and Jose, 1996).

In January 2001, a long-term study was initiated in the Mai Po reedbeds in co-operation with the Hong Kong Bird Banding Group, to develop a management strategy for the Mai Po reedbeds. This involves cutting and removing the reeds in four 1 ha experimental blocks on an annual rotation, and monitoring the use by birds of the different aged blocks through mist netting. A fifth block of reeds is maintained as a control block. The initial results show that the first block of reeds that was cut in January 2001 and has since regrown, attracted a greater diversity and abundance of birds after 1 year of growth, than compared with the other blocks that had not been cut for over 20 years.

One issue that has come up from this study so far, is that of the disposal of the 1 ha area of cut reeds. The initial block that was cut in January 2002 was burnt on-site after standard safety measures had been carried out. However, there was opposition to this as burning was seen as being "inappropriate" in a nature reserve. As a result, the second block of reed that was cut in January 2003 had to be removed off-site by volunteers and reserve staff, and would later be taken to a land-fill site for disposal. Discussion is continuing as to whether the block of reeds that will be cut in January 2004 will be burnt on-site or taken off site for disposal.

Lastly, management of the reeds at Mai Po also involves controlling their spread into open areas of water by either dredging or spraying with an approved herbicide (e.g. glyphosate; see Section: *Vegetation control*).

10.3.2. Gei wai Management

Gei wai Shrimp Harvesting. A wave of immigrants from China came to Hong Kong in the early 1940s, and they brought with them the idea of impounding the coastal mangrove forests to make intertidal shrimp ponds, locally known as *gei wai*. Although these *gei wai* were mainly managed for shrimp production, fish oysters, algae and brackish water sedges were also harvested.

Each *gei wai* has an area of approximately 10 ha and are examples of how coastal wetlands can be managed sustainably, i.e. so that they can be of benefit to local communities with minimal adverse impact to the environment. This is because traditional *gei wai* shrimp production relies on the natural productivity in the adjacent bay. The ponds are stocked by flushing in young shrimps from the bay in autumn, and the shrimps feed on organic matter, e.g. dead mangrove leaves on the bottom of the pond. As a result, the fishermen maintained the stands of mangroves inside the pond as a source of food for the shrimps and fish. The shrimp of main commercial importance is *Metapenaeus ensis* but fish, such as *Mugil cephalus* (Grey mullet) are also present.

Each *gei wai* has a single sluice gate that allows water exchange with Deep Bay via a channel through the coastal mangroves. The sluice gate is some 1.0–1.5 m wide, has concrete walls and wooden sluice boards which are slotted into grooves in the walls. Placing or removing the boards will prevent or allow water to flow through the sluice gate. The sluice boards have to be replaced every few years and the edges of any new boards have to be planed very carefully to ensure a watertight fit between boards.

Apart from a 10 m channel around the inner edge of each *gei wai*, there are also channels running the width of the pond to facilitate water exchange, and to allow a greater area for shrimp production.

Shrimp larvae are flushed into each *gei wai* from August-December on nights when there is a high tide in Deep Bay. The young shrimps feed on naturally occurring detritus on the *gei wai* floor. Shrimp harvesting takes place from the end of April until October or November and is done by opening the sluice gate when there is a low tide in Deep Bay, and placing a funnel net across the sluice gate to catch the outgoing shrimps. In the morning, water from Deep Bay is allowed back into the pond to maintain the water level, and to prevent heat stress which may cause the shrimps to die. Due to the high sediment load of the Pearl River, the water flushed into the *gei wai* from Deep Bay carries a high silt load. The sedimentation rate in the *gei wai* has been estimated to be 1.7 cm yr^{-1} (Lee, 1988). In order to maintain the channel at a suitable depth for shrimp production, dredging has to be conducted every 10 years.

After the end of the shrimp harvesting season, the *gei wai* are completely drained in turn for harvesting the fish inside. At this time, up to 1,600 wintering birds, such as herons, egrets and the endangered Black-faced Spoonbill *Platalea minor*, may be attracted into a single draining *gei wai* (Leader personal communication, 2000). There, they feed on the non-commercial fish and shrimp trapped in the pools of water at the bottom of the pond.

Ma (1997) studied the pattern of use of a draining *gei wai* by Little Egrets *Egretta garzetta*, and found that their numbers gradually increased as the pond was drained until numbers reached a peak on around the fourth or fifth day after draining began. Such peak in a single *gei wai* may represent as much as over 70% of the Little Egrets wintering in Deep Bay at that time (868 Little Egrets in *gei wai* 16/17 in November 1996).

Water Level Management. Under the Management Plan for the Mai Po Inner Deep Bay Ramsar Site (Anon., 1997) and the Mai Po Management Plan (Young, 1999), only *gei wai* 12–14 are to be operated as traditional *gei wai* shrimp ponds. The other *gei wai* are managed for wildlife and the ecological habitats (e.g. reedbed) that they may support.

Management of the *gei wai* water level for shrimp production has already been discussed in the previous section. For the other ponds, water level management depends on the conservation objective of that pond. For example, the management objective for *gei wai* #11 and 16/17, is to provide a high-tide roosting site for migratory shorebirds during spring and autumn passage. As a result, the water level is lowered during these two seasons. During summer when few shorebirds are present, the water level is raised so as to prevent vegetation growth on the areas of exposed mud. It is also to prevent the encroachment by reeds and other vegetation from the edge of the pond into the central open areas of water. In the case of *gei wai* #16/17 the water level is kept high during winter, so this pond acts as a night-time roosting site for wintering waterfowl.

The *gei wai* bunds occasionally leak, and if these are small they can be repaired by hand. Sometimes, a large section of the bund may collapse because of wave action during storms, and repairs would then have to be undertaken by machinery, such as a backhoe digger or a dredger on pontoons.

10.3.3. Freshwater Pond Management

Prior to the mid-1970s, there was a large freshwater marsh at the site of a present housing estate adjacent to Mai Po, which was known to support the last breeding population of Pheasant-tailed Jacanas *Hydrophasianus chirurgus* in Hong Kong (Carey et al., 2001). As this marsh has now been lost, it was recommended in the Management Plan for the Ramsar Site (Anon., 1997) and the Mai Po Management Plan (Young, 1999), that a series of freshwater marshes be established at the southern end of MPNR (*gei wai* #20–24).

Work on setting up such a freshwater marsh began in 1997 in *gei wai* #20. This *gei wai* consist of six ponds which were commercially operated as fish ponds until 1995, when the government resumed the land and handed it over to WWF Hong Kong for conservation management. The ponds were first drained to remove the brackish water inside and to repair any leaks in the bunds using a bulldozer and backhoe. As one of the objectives of the ponds was to attract amphibians and odonates, fish were not restocked into the ponds as these may be potential predators. After refilling with rainwater, a number of the ponds soon developed a plant community which was previously uncommon at Mai Po, with species such as the grasses *Paspalum distichum* and *Echinochloa crus-galli*, which are a food source for a number of grazing and granivorous waterbirds respectively. At nights, these ponds were able to attract large number of wintering waterfowl that would return to roost, especially amongst the grasses. In summer, these rain-fed ponds supported a large number of odonates (Young, 2001).

As these rain-fed ponds were created for the first time at Mai Po, learning how to manage these ponds for wildlife was initially very much on a trial-and-error basis. Just 1–2 years after the ponds had been created, it was noticed that the grasses which the waterfowl were often seen grazing and roosting amongst, were being invaded by weeds and climbers. As a result, a programme was initiated to control the problem. Another problem was that after 3–4 years, Catfish *Clarias fuscus* colonised the pond, and began to graze out the grasses that the waterfowl themselves grazed and roosted amongst. To resolve this problem, the ponds had to be drained and the fish removed.

10.3.4. Mangroves

The Deep Bay mudflat is the main feeding area for up to 68,000 migratory waterbirds that either spend the winter in the Bay, or pass through on migration in spring and autumn. Recent studies have found over 80 species of polychaete worms, snails, bivalves, crabs, mudskippers etc. on the mudflat (McChesney, 1997), with about 20 species that were either new to science or new to Hong Kong (Lee, 1993).

It is a natural part of succession for coastal mudflats to silt up and increase in height, whilst the vegetation on the landward side of the mudflats, mangroves in the case of the Deep Bay mudflats, to slowly extend out over the mudflats as it silts up. However, ^{210}Pb analysis of sediment cores from Inner Deep Bay has shown that the rate of sedimentation may have doubled since the mid-1980s (Peking University, 1995), probably due to a combination of an increase in the sediment load in the water and reduction in water flow through the Bay. The current sedimentation rate is estimated to be some 1.3–2.8 cm per year (Ove Arup, 2002).

The short term effect of the increase in sedimentation, is that mangrove seedlings (mainly *Kandelia candel*) which colonise the mudflat in front of the floating hides on the edge of the Mai Po mangroves, have to be removed on an annual basis. This is so as to maintain an area of mudflat for feeding waterbirds, also so that the mangrove trees do not obstruct the view from the hides as they grow.

In view of the fact that the mudflats do not appear to be expanding out into Deep Bay (Fung pers. comm.; Chinese University of Hong Kong), colonisation of the mudflat by mangroves over the long-term will lead to the loss of the mudflat feeding habitat for waterbirds. This problem still has to be addressed by the Hong Kong SAR Government, and a management strategy developed for managing the Deep Bay mangrove.

10.3.5. Visitor Management

In order to ensure the long-term protection of wetland reserves such as Mai Po, there is a need to promote public awareness of the importance of the site, such as through guided visits. However, there is a need at the same time, to minimise the disturbance caused through large numbers of such visits. As a result, visitor access into any protected area has to be controlled. Whilst access to some parts of the reserve may be unrestricted, access to other parts may have to be restricted, depending on the importance of these areas for wildlife and their sensitivity to human disturbance.

In 1976, it was decided that with the increasing urbanisation of the area around Mai Po, access to the site would have to be restricted to those people holding Mai Po Entry Permits issued by the Agriculture, Fisheries and Conservation Department (AFCD). These permits are issued to members of the Hong Kong Birdwatching Society, students or researchers, or volunteers working at Mai Po.

The restricted access status of the MPNR is enforced by AFCD, who have a Nature Wardens office by the main entrance into MPNR. All visitors to the reserve must first show the Nature Wardens their valid Mai Po Entry Permit before being allowed in. Anyone who enters the reserve without such a permit may be liable to a fine.

Groups of visitors to Mai Po are mainly guided along the eastern, landward portion of the reserve from the car park to the Wildlife Education Centre and a Nature Trail. Certain groups of secondary school students will have the additional opportunity of visiting the Floating Boardwalk and Hides. Generally, groups of general visitors and students are not taken to the southern *gei wai* (*gei wai* #20–24), which is managed as a relatively undisturbed part of the reserve.

Some 40,000 people visit Mai Po each year, with about one quarter being school students on one of the 400 special visits organised for primary and secondary students. The Education Department funds the school visits and outside charities sponsor visits by the disabled. However, visitors on the public visits have to make a minimal donation to WWF Hong Kong.

10.4. Summary

From the early 1990s, there has been an increase in public awareness of the importance of environmental conservation which has lead to increased numbers of people visiting the MPNR. Apart from learning about the ecological and cultural importance of the Reserve, many visitors, especially birdwatchers, have also taken an interest in the management of the Reserve itself. Whilst such interest is to be promoted, there is also a need for the Reserve to provide a better explanation of how the site is managed so that apparent conflicts can be avoided, such as why:

- access to the Reserve has to be restricted;
- management tools such as herbicides and fire have to be used;
- trees have to be felled from within a nature reserve; and
- mangrove trees have to be removed from the mudflat.

Through greater public awareness of the work of the reserve, it is hoped that there can be greater support for the reserve's work.

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Chapter 11

Conservation and Uses of Mangroves in Hong Kong and Mainland China

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Abstract. Mangroves are important inter-tidal wetlands on tropical and subtropical coasts and have been seriously damaged. In Mainland China and Hong Kong SAR, mangrove resources have been rapidly destroyed in recent decades due to massive reclamation, infra-structural developments and pollution. Information related to conservation and uses of mangroves in China is often scattered and incomplete. The present chapter aims to review the distribution, functions and uses of mangrove resources, the conservation strategies, and the associated problems in China.

11.1. Introduction

Mangrove ecosystem is found in inter-tidal areas of sheltered coastlines between 35°N and 35°S. The ecosystem includes unique salt-tolerant trees, shrubs, and other organisms with special adaptation to fluctuating water levels which create a stressful environment. The mangrove ecosystem connects terrestrial, freshwater and marine ecosystems together, and is considered as one of the world's most productive ecosystems with high levels of primary productivity (Kaplowitz, 2001). The living biomass and litter production of mangrove forests are very high (Table 1). The litter has rapid decomposition and nutrient cycling, thus exporting large amounts of plant debris (organic matter and nutrients) to outside water, initiating mangrove-derived detritus food chains and food webs, and supporting fisheries and other aquatic production. In addition, mangroves provide habitats for valuable plant and animal species.

On the other hand, the mangrove ecosystem is among the world's most threatened ecosystems, with more than half of the original area already lost in the most heavily impacted areas (Field et al., 1998). For instance, mangrove cover in the Philippines was reduced from 288,035 ha in 1970 to 123,400 ha in 1993, largely as a result of aquaculture development. Increased market integration,

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Table 1: Annual above-ground biomass, total biomass, and litter production (t ha^{-1}) of mangrove forests in China and East Asia.

Country	Region	Mangrove species	Above-ground biomass	Total biomass	Litter biomass
China	Mai Po, HKSAR	*Kandel candel*	129.6	NA	11.07
	Futian, Shenzhen	*Aegiceras corniculatum* and *K. candel*	87.1	121.4	11.69
		Sonneratia apetala and *S. caseolaris*	33.9	45.6	12.18
	Dongzhai, Hainan	*Bruguiera sexangula*	248.5	420.3	11.79
	Hegang, Hainan	*B. sexangula*	NA	NA	12.55
	Wenchang, Hainan	*B. gymnorrhiza*	64.1	91.5	NA
	Jiulongjiang, Fujian	*K. candel*	93.4	162.6	9.21
	Yingluo, Guangxi	*Rhizophora stylosa*	196.2	291.6	6.32
	Beihai, Guangxi	*Avicennia marina*	26.9	52.7	NA
	Danshui, Taiwan	*K. candel*	NA	143.9	NA
Thailand	South Thailand	*R. apiculata*	159.0	NA	6.70
Malaysia	Matang	*R. apiculata*	185.3	209.5	9.71
Philippines	NA	Mixed mangroves	45.9	NA	NA

NA, not available. Lee (1990), Tam et al. (1995a, 1998), Lin (1999), and Wang et al. (2002).

modernization of traditional economies and urban development in recent decades had led to more intensive mangrove exploitation, and even to their removal (Gilbert & Janssen, 1998). In Hong Kong SAR and southeastern coasts of China, mangroves are destroyed due to urbanization, agriculture and aquaculture. China's mangrove coverage has plummeted, from 50,000 ha in the late 1950s to 15,000 ha in year 2000, and currently occupies less than 0.1% of the world's total. Underestimation of the total value and of the impacts of human activities is another major factor contributing to the widespread loss and degradation of mangrove ecosystems. Therefore, it is important to understand the functions and uses of mangrove resources, and compromise the needs of various stakeholders prior to development of a conservation and management plan. The chapter aims to review the mangrove distribution, its uses, conservation, and associated problems in China.

11.2. Mangrove Distribution and Characteristics in Hong Kong and Mainland China

Mangroves in China are distributed in five provinces, namely Guangdong, Guangxi, Fujian, Zhejiang and Hainan, in addition to Taiwan, Hong Kong and Macau Special Administrative Regions (Table 2). The mangroves in Zhejiang do not occur naturally, and the remaining 8 ha of *Kandelia candel* were transplanted in the 1950s (Li & Lee, 1997). The current mangrove coverage is around 15,000 ha, around 0.2% of South and Southeast Asia mangroves. Over 80% of the existing mangroves are secondary forests with an average height of 1–2 m. A total of 26 true mangrove species and 11 associate mangrove species are found in China, representing about 45% of all mangrove species (83 in total) in the world. In China, Hainan has the highest mangrove species richness and the best developed mangrove forests, while Guangdong has 11 true mangrove species and 9 associate mangroves (Table 3). Mangrove plants such as *Sonneratia paracaseloris*, *S. hainanensis*, *Lumnitzera littorea* and *Nypa fruticans* have become rare and endangered species.

The mangrove ecosystems in China also support a diverse group of macrobenthic animals (Table 4) and provide food for other wildlife such as birds. Many diversified species of precious birds have been identified. Around 201 species of birds belonging to 39 families and 17 orders have been recorded in mangroves in China; 83 species are reproductive resident or summer-migrants, and 118 species are traveling or winter-migratory birds (Lin, 1999). The mangrove nature reserves, in particular Mai Po in HKSAR and Futian in Shenzhen, are important stopovers for winter migratory birds flying from Australia to Siberia. Each year, more than 100,000 migrants flying to the southern areas have been recorded in Futian mangroves (Wang et al., 2002).

Table 2: Mangrove Nature Reserves in China.

Region	Name	Time established	Protected area (ha)	Mangrove area (ha)	True mangrove species no.
Hainan	Donzhaigang[N]	1980	5240	1760	20
	Qinglangang[N]	1981	2948	2722	26
	Xingyin[C]	1983	67	NA	12
	Huachang Bay[C]	1984	133	NA	12
	Caiqiao[C]	1986	350	NA	12
	Xingying[C]	1986	133	NA	16
	Qingmeigang[C]	1989	156	30	20
Guangdong	Futian[N]	1984	304	111	7
	Zhanjiang[P]	1991	2000	933	11
	Zhuhai[S]	Plan	ND	60	5
Guangxi	Beilunhe[P]	1990	2680	1207	11
	Shankou[N]	1990	4400	730	12
Fujian	Jiulongjiang[P]	1988	200	108	5
Taiwan	Danshui[P]	1985	733	80	1
HK SAR	Mai Po	1976	380	120	8
Macau SAR		No protection	<1	0	5

NA, data not available; N, national level; P, provincial level and C, local county level; S, suggested to protect. Li & Lee (1997) and Lin & Fu (1995).

11.3. Uses and Functions of Mangroves

11.3.1. Uses of Mangroves in the World

Table 5 summarizes the possible uses of mangrove resources. It is obvious that mangroves play an important role in maintaining a healthy coastal ecosystem by exporting large quantities of detritus and supplying abundant food and feed to aquatic organisms, maintaining nutrient cycling, biogeochemical functions and energy flow along complex food chains and food webs. It also provides a habitat for a variety of animals especially waterfowl and wintering birds, and acts as a nursery for juvenile species through provision of food and shelter from predation. The prop roots and pneumatophores of mangrove plants, and the shading effect under the leaf canopy allow the small animals escape or hide from their predators.

In addition to ecological functions, mangrove ecosystems are important to the subsistence livelihoods of tropical coastal communities (Kaplowitz, 2001). At present, millions of coastal dwellers throughout the region are dependent on

Table 3: True mangrove species in Guangdong Province, China.

Family	Species	Zhanjiang	Zhuhai	Futian	HKSAR	Macau
Rhizophoraceae	*Bruguiera gymnorrhiza*	+	+	+	+	+
	Ceriops tagal	+	−	−	−	−
	Kandelia candel	+	+	+	+	+
	Rhizophora stylosa	+	−	−	−	−
Acanthaceae	*Acanthus ebractearas*	+	−	−	−	+
	Ac. ilicifolius	+	+	+	+	+
Combretaceae	*Lumnitzera racemosa*	+	−	+	+	+
Euphorbiaceae	*Excoecaria agallocha*	+	+	+	+	−
Myrsinaceae	*Aegiceras corniculatum*	+	+	+	+	+
Sterculiaceae	*Heritiera littoralis*	+	−	+	+	−
Avicenniacea	*Avicennia marina*	+	−	+	+	+
Total		11	5	8	8	7

mangroves for their livelihoods. Mangrove ecosystems can be directly exploited by extracting fish, agricultural products, and wildlife, as well as a variety of other goods including wood for fuel, construction and building materials, drugs, chemicals, feed and food (Kovacs, 1999). Mangrove ecosystems and their ecological functions also provide an array of important indirect services for people such as prevention of storm damage, flood and water control, support of fisheries, pollution mitigation, recreation and transport.

Positive correlations between mangrove areas and near-shore fish and shrimp catches have been demonstrated in many countries including the Philippines, Malaysia, Indonesia, and Australia, with an annual fisheries related income ranging from US$ 66 to almost US$ 3,000 per ha of mangrove (Baran & Hambrey, 1998). In Southeast Asian countries like Thailand and Indonesia, the estimated annual income from fisheries, forestry and agriculture was around US$117–130, $30–67, and $165 per ha of mangroves, respectively (Gilbert & Janssen, 1998).

Table 4: Benthic macrofauna in China mangroves in terms of number of species in each group, density (number of individuals m^{-2}) and biomass ($g\ m^{-2}$).

Family	**Dongzhai, Hainan**	**Futian, Shenzhen**	**Hong Kong SAR**	**Longhai, Fujian**
Annelida: Polychaetes	29	5	2	41
Mollusca: gastropods and bivalves	51	37	52	51
Crustacea: shrimps and crabs	32	27	26	57
Osteichthyes: fishes and mudskippers	4	11	3	11
Other animals	4	4	17	12
Total species no.	138	84	100	172
Density	249	55–1,305	24 (5–44)	311
Biomass	150	29–78	116	48
References	Jiang et al. (1997)	Yu et al. (1997)	Cai et al. (1997); Tam & Wong (2000a)	Jiang & Li (1995)

In the Philippines, the estimated net annual economic values of wood and fish products from managed mangroves were US$90 and $538 per ha mangroves, respectively (De Leon & White, 1997). The income generated from mangrove wood production and agriculture is poorly understood as it varies from species to species, and the best species for local uses is often not clear due to lack of systematic studies (Kovacs, 1999).

The estimated values of the services derived from mangroves (e.g. shoreline protection against erosion, food and wood production, and habitat for wildlife) at US$10,000 per ha per year were reported by Costanza et al. (1997). Mangrove wetlands have long been used as convenient sites of waste disposal and have often received untreated wastewater from human activities (Clough et al., 1983). The feasibility of using mangroves to remove pollutants from municipal sewage, livestock wastewater and shrimp effluent has been examined since 1990s (Robertson & Phillips, 1995; Wong et al., 1997; Rivera-Monroy et al., 1999; Tam & Wong, 1999). Integrated pond–mangrove farming systems with wetland to shrimp pond ratios varying from 1:2 to 1:22 have been proposed; depending on the capacity of mangrove sediments to remove nitrogen relative to the anticipated loading rate (Robertson & Phillips, 1995) the ratio could be lowered to a range of 1:0.04–1:0.12 if denitrification process takes place in the mangrove wetland

Table 5: Uses and functions of mangroves.

Uses	Examples
Ecological	Maintain functions of ecosystem: nutrient cycling and biogeochemical function Support biodiversity Provide wildlife habitat and nursery for juveniles and post-larvae
Consumptive use for harvestable products	Mangrove woods: timber, charcoal and fuel wood, construction and building Fisheries and aquaculture: fishes, shrimps, *gei wai* prawns, snails, and crab cultivation and collection Farming: paddy and rice fields Salt extraction Tannin extraction Medicine: Chinese herbs Bioactive compounds and chemicals extraction
Non-consumptive uses	Recreation: ecotourism, picnics and aesthetic values Storm protection: prevent erosion, flood and water control Pollution mitigation: wastewater treatment, waste adsorption and absorption Education, training, and scientific research

system (Rivera-Monroy et al., 1999). The mangrove sediments are effective in retaining nutrients and heavy metals from wastewater, and the estimated amounts of nitrogen and phosphorus removed from a mangrove system ranged from 0.18–87.6 mg N m^{-2} day^{-1} and 2.7–9.9 mg P m^{-2} day^{-1}, respectively (Clough et al., 1983; Chen et al., 1995; Robertson & Phillips, 1995; Tam & Wong, 1996).

11.3.2. Functions of Mangroves in China

In history, mangroves were of great value in China and almost all uses listed in Table 5 have been practiced. Mangrove woods have been used to make furniture and small boats (Lin, 1999). However, due to the relatively small sizes of mangroves in China, and the lack of planned reforestation, mangroves are not suitable for economic uses as wood and fuel resources. Seeds, fruits and hypocotyls have been consumed by humans especially in early 1960s. The hypocotyls of *Bruguiera* spp., *Rhizophora* spp. and *K. candel*, containing high concentrations of starch, have been sliced and ground into powder for making cakes or used as sweetened stuffing for pastry, while salted seeds and fruits of *A. marina* and *S. caseolaris* have been eaten for breakfast and banquets (Lu & Lin, 1987) after being soaked in water for several hours to rinse out tannin substances. The mangrove plants in China have been widely used as traditional herbal medicines (Table 6) but very poorly documented historically, and much information is based on conventional wisdom and anecdote. There is no research or scientific proof for their effectiveness. The food/feed and medicinal potential of mangrove plants deserves further research and development work.

Aquaculture has been practiced in China for many years. The mangrove benthic species with high commercial values including *Sipunculus nudus* (worm), *Phascolosoma esculenta* (known as To Sun Dong, a traditional local snack in Xiamen), *Pinctada martensi* (pearl-mother shell for pearls), *Meretrix meretrix* (shell), *Saccostrea cucullata* (oyster), *Scylla serrata* (crab), *Metapenaeus monoceros* (shrimp), *Bostrichthys chinensis* (fish), and various species of mullets,

Table 6: Medicinal uses of true mangrove plants in Guangdong, China.

Species	**Plant part**	**Medicinal uses**
Kandelia candel	Root	Chronic arthritis
Bruguiera gymnorrhiza	Fruit and hypocotyl	Diarrhea, malaria and diabetes
	Leaf	Malaria
Rhizophora stylosa	Bark	Purify bloody urine
Ceriops tagal	Bark, seed and leaf	Hemostasis, astringent and scabies
Avicennia marina	Leaf	Skin abscess, diuretic
	Trunk and bark	Contraceptives, diuretic
	Fruit	Diabetes
Acanthus ebracteras	Fruit	Furuncle
Acanthus ilicifolius	All parts	Analgesic, swelling relief, leukemia, cancer, skin itches and abscess
	Leaf	Rheumatic pain
	Fruit and root	Snake wound, hepatitis B, impotence (male sterility)
Lumnitzera racemosa	Juice of trunk	Aphtha
	Bark	Diabetes, kidney stone
Excoecaria agallocha	Bark	Diarrhea
	Leaf	Epilepsy
	Wood burning	Leprosy
Heritiera littoralis	Bark	Haematuria, diarrhea

Lin (1999) and Lin & Fu (1995).

mudskippers and sand borers have been cultivated and give economic benefits to the local community (Lin, 1999). Lin & Fu (1995) estimated that the yearly output of *Scylla serrata* in Hainan Donzhaigang mangroves was more than 30,000 kg.

In recent years, more efforts have been placed to explore the aesthetic values of mangroves and to promote ecotourism. People can do canoeing, fishing, picnicking, observing the behavior of birds and animals, and understanding the adaptation of plants in mangrove forests. Tourists can appreciate the beauty of nature and simultaneously understand the ecosystems and biodiversity, which can enhance their awareness on mangrove conservation. Mai Po and Futian Mangrove Nature Reserves have been successful in developing education and research programs, school visits and guided tours. In 1997, over 40,000 students and public visited Mai Po Nature Reserve without compromising its conservation value. The Dongzhaigang Mangrove Nature Reserve in Hainan is the largest tourist zone for mangrove sightseeing in China, and has attracted more than 200,000 visitors since 1980 (Lin & Fu, 1995). Other mangroves such as Shankou National Nature Reserve in Hepu, Guangxi and Qinglangang Mangrove Nature Reserve in Hainan are also planned for mangrove ecotourism. Although tourism gives economic benefits to the local community and brings additional funds for mangrove conservation, it also generates damages and disturbances, in particular, the need for more food and accommodation facilities, and pollution generating from human activities. How to compromise these conflicts and achieve real ecotourism still needs further research. The carrying capacity of a mangrove environment must be properly estimated and the potential damage must be understood before converting China mangroves into tourist spots.

11.4. Conservation of Mangroves

11.4.1. General Principles

The regional and local losses of mangroves have attracted more and more attention, and emerging awareness of the societal costs has encouraged a recent trend towards preservation and restoration of mangroves. The importance of mangroves started to be realized in 1978 when UNESCO's Scientific Committee on Oceanic Research established a working group on mangrove ecology. The mangrove ecosystem was identified as an endangered ecosystem in 1980s, and a working group on mangrove ecosystems was created by IUCN (International Union for the Conservation of Nature) (Ellison & Farnsworth, 1996). Many conferences have been convened by UNESCO, UNDP and UNEP, and have formulated "Charter for Mangroves" with ISME (the International Society for Mangrove Ecosystem) for further understanding on each country's mangrove features and related protection measures.

For better conservation and management of mangrove ecosystems in a sustainable manner, it is essential to have a clear understanding of the features and ecological functions of mangroves in local contexts to properly estimate the uses and values of mangroves, and to address the needs and interests of all stakeholders in the system. In a mangrove ecosystem, due to its complexity and multiple ecological and socio-economic functions (Tables 5 and 6), various stakeholders are involved. These include (a) primary stakeholders: local communities whose livelihoods are directly dependent on mangrove resources such as fishermen, paddy farmers, wood makers, and others; (b) key stakeholders: government officials, developers, and resource economists whose actions directly affect decision-making; (c) secondary stakeholders: tourists and traders who have an interest on the mangroves but without any direct involvement; and (d) ecologists and conservationists who have strong view to preserve the nature of the mangroves. Proper trade-offs and negotiation, to balance the gains and losses amongst all stakeholders, are essential processes to solve conflicts before decision-making. The arrays of benefits flowing from mangrove ecosystems perceived at the local level are important considerations. A success and sustainable conservation plan will rely on how these views on mangrove functions and values are cared for and understood.

In view of the fact that mangroves often cover a large area and are scattered in different sites within a region, and that resources and manpower are always limited, it is important to prioritize sites worthy of conservation. These sites can only be identified and selected after taking into account of all available and relevant scientific, social, economic and cultural values. However, detailed information relating to the distribution of the biota to be protected, the socio-economic data, the degree of pollution and human disturbance is largely unavailable, and obtaining such data is often expensive and time consuming.

11.4.2. Conservation of Mangroves in HKSAR

In Hong Kong SAR, mangroves were used in the past for production of salts, shrimps, and other fisheries, but they are not used for any commercial production at present. They are mainly conserved to maintain the natural functions and processes of the ecosystem, biodiversity, and habitats for birds and wildlife. They are scattered in more than 40 different enclosed bays, in addition to the largest area in Mai Po Nature Reserve. In order to assess and prioritize the conservation values of these scattered mangroves, a comprehensive study was conducted in 1994–1997 to investigate the distribution, ecological and socio-economic characteristics of 43 mangrove swamps still remaining in Hong Kong (Tam et al., 1997; Tam & Wong, 2000a, 2002). Nine criteria covering ecological, economic and social aspects were use to develop a conservation score for each mangrove swamp, and the swamps were classified into five categories according to their scores. The extremely important swamps have been conserved immediately (Tam & Wong, 2002). Among the 10 conservation strategies recommended, enforcement of existing ordinances and legislation, environmental education and community actions to promote public awareness, and replanting are found to be the most effective and suitable measures in HKSAR, and have been adopted by local government (Tam & Wong, 2000a, 2002).

11.4.3. Conservation of Mangroves in Mainland China

As early as in the 19th century, the Chinese had recognized the importance of mangroves, and penalized people who caused damage, while rewarding those who planted mangroves (Field, 1996). In the 1950s, the total mangrove area in China was around 50,000 ha, but more than 30,000 ha of mangrove, with an annual loss of 600 ha, have disappeared in the last 50 years (Table 7). Since 1980s, China has started to protect mangroves from disturbance by establishing mangrove nature reserves. Since then, 14 mangrove nature reserves including Mai Po in Hong Kong SAR and Danshui River in Taiwan have been established to protect around 8,000 ha of mangrove, around 44% of existing mangroves in China (Table 2).

Table 7: Damages of mangroves in China in 1950–2000.

Year	**Region**	**Area of damages (ha)**	**Reasons**
1960s–1970s	Fujian, Guangdong and Guangxi	Massive	Conversion to farmland for crop production
	Hainan Island	4,667	Conversion to rice fields
1980s	Fujian	470	Enclosed as ponds for cultivation of prawns, crabs and eels
	Guangdong	13,972	Conversion to shrimp and prawn farms
Mid 1980s onwards	Guangdong	Massive	Urban and infra-structural development
	Futian	48	Infra-structural development
	Zhuhai	1,350	Urban and infra-structural development, and pollution

Modified from Lin & Fan (1992) and Wang et al. (2002), and personal communication.

In each nature reserve, three zones, namely core, buffer and experimental (research) are established. For instance, the Futian National Nature Reserve established in 1984 had a total area of 415 ha, and the core, buffer and experimental zones occupied 203, 65 and 37 ha, respectively. In recent years, due to infra-structural developments in Shenzhen, the area of the Futian nature reserve has been reduced to 367 ha with 44, 14 and 39% as core, buffer and experimental zones, respectively (Wang et al., 2002). Within the core zone of each nature reserve, utilization and interference from external factors are restricted, and the mangroves are undisturbed and left in their natural state in order to maintain the biodiversity, ecological processes and functions. In the buffer zone, ecological tourism and sustainable utilization of mangroves for aquaculture (e.g. *gei wai* ponds in Futian, Shenzhen; oyster culture in Shankou, Guangxi; Pearl Culture in Dongzhaigang, Hainan) and other purposes are carried out through appropriate management plans. Research, education, training and community action programs are taken place in the research and experimental zone. The goals and objectives of each nature reserve vary slightly (Table 8).

11.4.4. Mangrove Planting and Restoration

In view of the significant loss of mangrove habitats in past decades, one of the important conservation measures is to restore mangrove habitats by planting and replanting. Technologies required for mangrove replanting and restoration are neither new nor complex. Several hundred mangrove restoration projects have been undertaken around the world, initially confined to First World and Caribbean nations, and have recently been spread to South and East Asia (Field, 1996). Mangroves have been planted as part of a forestry management regime and as a coastal protection measure in countries such as Thailand, Malaysia, Indonesia, the Philippines and Bangladesh. However, most of these are economic plantation projects intended to yield firewood, construction timber, or other forestry products for livelihood improvement of the local community. The results often resemble monotypic, even-age stands cultivated by commercial forestry operators, with little consideration on ecological restoration.

In China, in contrast to other countries in East Asia, economic planting for direct extraction of mangrove products is not the major objective, and large-scale planting projects for woods are uncommon. Instead, more consideration has been placed on enhancing the ecological values of the mangroves, i.e. preservation. In most regions in China including Hong Kong SAR, the inter-tidal zones, compared to those in tropical regions, are relatively narrow which limits the mangrove coverage. Recent reclamation and urban development projects further destroy mangroves at the seafront. The mangrove coverage at present often exists as a thin

Table 8: Major objectives of mangrove conservation in some nature reserves in China.

Region	**Name**	**Protection level**	**Conservation objectives**
HKSAR	Mai Po	Part of RAMSAR	Ecological habitat for biodiversity; stopover for migratory birds; water fowling and wildlife; education, training and scientific research
Shenzhen	Futian	National Nature Reserve	Ecological habitat; important stopover for migratory birds; ecotourism; education, scientific and applied research
Guangxi	Shankou	National Nature Reserve	Ecological habitat and biodiversity; pearl oyster culture for "Southern Pearls"; develop offshore fishery; protecting sea shore and terrestrial resources; scientific research
Hainan	Dongzhaigang	National Nature Reserve and RAMSAR	Protect wetland habitat and biodiversity; develop sustainable ecotourism and related business; education and scientific research
Fujian	Jiulongjiang (Longhai)	Provincial Nature Reserve	Ecological habitat; protect sea shore from erosion; historic attraction; scientific research

belt (<40 m) with relatively low ecological and economic values. Therefore, the main aim of planting is to increase the mangrove extent, and create a "green wall" along coastlines. A successful ecological planting project should aim for multiple species enrichment planting in order to increase both the extent and biodiversity of the planted forests. However, planting in Hong Kong SAR and Mainland China is often limited to viviparous species, with obvious droppers such as *Kandelia candel* or *Bruguiera gymnorrhiza* (Tables 9 and 10). In recent years, planting in Mainland China has concentrated on rapid growing of the exotic pioneer species, *Sonneratia*, as planting native species such as *Aegiceras corniculatum* and

Table 9: Mangrove planting projects in Hong Kong SAR.

Place	**Year**	**Area/no. of plants**	**Species planted**	**Purposes**	**Organization**
Sheung Pak Nai, Deep Bay	1980s–present	4 km along coastlines	*K. candel*, *B. gymnorrhiza*, and *A. corniculatum*	Protect fish ponds from erosion	Local farmers
Tin Shui Wai Creek, Northwest New Territories	1994	10m wide, 20–100m long along the creek	*K. candel*	Compensate loss due to Tin Shui Wai New Town reclamation	Territory Development Department, HK Government
Yuen Long and Kam Tin, Northwest New Territories	1995	34 ha	*K. candel*, *B. gymnorrhiza*, *A. corniculatum*, and *Ac. ilicifolius*	Mitigation for loss along drainage channels	Territory Development Department, HK Government
Wong Chuk Wan, Sai Kung	1995–present	80,000 droppers	*K. candel* and *B. gymnorrhiza*	Green action	Friends of Earth, HK (FOE)
Kei Ling Ha Lo Wai, Sai Kung	1999–present	No record	*K. candel*, *B. gymnorrhiza*, and *A. corniculatum*	Green and community action	FOE, Ocean Park Conservation Foundation
Kau Sai Chau	1995–1997	1.64 ha	*K. candel*, *B. gymnorrhiza*, and *A. corniculatum*	Compensate loss due to public golf construction	Jockey Club, HK
Tai O	Plan to start in 2005	12 ha	Multiple and mixed species	Mitigation for loss due to Chek Lap Kok Airport construction	HKSAR Governments

Table 10: Mangrove planting and restoration projects in Mainland China.

Place	Year	Area/no. of plants	Species planted	Purposes	Organization
Qinzhou, Guangxi	1956–1965	7 ha	*Av. marina*	Animal feed	Local farmers
Haikang, Guangdong	1956–1965	100 ha	*R. stylosa*	Protect land farm, fish ponds and houses	Local people
Longhai, Fujian	1958	No record	*K. candel*	Protect erosion	Local people
Dongzhaigang, Hainan	1980–1990	173 ha	*K. candel*, *B. gymnorrhiza*, *B. sexangula*, *X. granatum*, *R. apiculata,* and *R. stylosa*	Conservation	Nature Reserve
Futian, Shenzhen	1986–1998	>50 ha	*K. candel*, *A. corniculatum*, *B. sexangula*, *S. caseolaris*, *S. apetala*	Expand the mangrove belt, and conservation	Nature Reserve

(*continued*)

Table 10: Continued.

Place	**Year**	**Area/no. of plants**	**Species planted**	**Purposes**	**Organization**
Zhanjiang, Leizhou Peninsula, Guangdong	1985–1994	>1,200 ha	*A. marina*, *R. stylosa*, *K. candel*, and *B. gymnorrhiza*	Conservation	Nature Reserve
	2001–present	50 ha	*S. apetala*	Create a green belt	Local government
Qinzhou, Guangxi	1982–1984	1,400 ha	*A. corniculatum*	Conservation	Local government
Xiamen, Fujian	1996–2001	107 ha along 2.3 km coastlines	*K. candel*	Create a green belt	Local government
Zhuhai, Guangdong	1999–present	140 ha	*S. apetala*, *R. stylosa*, *B. gymnorrhiza*, and *H. littoralis*	Create a green belt	Local government
Dongxin, Guangxi	2000	600 ha	Not record	Conservation	Local government
Dazhou Island, Maoming, Guangdong	2001–2002	660 ha	Not record	Develop a sea forest park	Local government
Punyu, Guangdong	2002	15 m wide × 15 m long	*S. apetala*	Conservation	Local government

Avicennia marina has failed most times. More research is needed to develop planting methods for non-viviparious and rare species, and to understand the long-term impacts of exotic mangrove species. It is also important to create nurseries for research, and the continuing supply of mangrove propagules for planting and reforestation.

11.5. Problems and Possible Solutions in Mangrove Conservation

11.5.1. Habitat Loss Due to Land-Use Change

Land-use change leading to habitat loss is one of the major threats to mangroves. As shown in Table 7, mangroves in China were mainly destroyed due to conversion of mangroves to salt pans, *gei wai* ponds, shrimp and crab farms, fisheries and paddy fields during the period 1950–1980s. In recent decades, urbanization, reclamation and infra-structural developments have destroyed most mangroves along coastlines. In Shenzhen, 36 ha of mangroves have been destroyed since 1991 (Wang et al., 2002). In Hong Kong SAR, 7 ha of mangroves were lost due to construction of Chek Lap Kok Airport, and 85 and 42% of the original mangrove cover in Deep Bay and Tolo Harbor were lost since the 1970s, respectively (Tam et al., 1997). It is obvious that mangrove preservation requires land use planning decisions. In Hong Kong SAR, important mangrove areas have been included as Country or Marine Parks, Sites of Special Scientific Interests (SSSIs) and Coastal Protection Areas on relevant Outline Zoning Plans (OZPs) or Development Permission Area Plans (DPA). Any development taking place in areas having mangroves of high conservation values will require planning permission from the Town Planning Board, and prevention, minimization, mitigation and compensation measures for mangroves, based on ecological impact assessment, must be included. In the Mai Po and Inner Deep Bay RAMSR site, planning control in the buffer zone is enforced to discourage developments, and the concept of no net loss of wetlands has been adopted by the Town Planning Board, Hong Kong SAR Government since 1999.

11.5.2. Water Pollution and Human Disturbance

Water pollution is another important threat to mangroves in both HKSAR and Mainland China as mangroves are located in sheltered and relatively slow-flushing bays. Deep Bay with increased pollution from Shenzhen River and other local feeders is famous for its pollution, mainly due to discharges of untreated or partially treated wastewater from cities and villages in HKSAR and Shenzhen Special Economic Zone (Richardson et al., 2000). Table 11 summarizes the degree of pollution in HKSAR, Shenzhen and other mangroves. Pollution loads arise from urban, industrial and agricultural sources within the mangrove areas, and the catchments must be reduced and restricted. The governments should make efforts to install or upgrade sewage treatment plants to keep pace with the rapid economic development in the areas, tighten the discharge standards, and take enforcement actions on industrial and agricultural illegal discharges. Other human disturbances, such as illegal cutting, refuse dumping, eutrophication and algal blooms, and accidental discharge of toxic pollutants, which have also killed mangrove plants, must be controlled.

11.5.3. Lack of Ecological and Baseline Data

One problem in conserving mangroves is lack of understanding regarding its ecological features. A long-term monitoring program aiming to quantify baseline conditions of the area, and to detect any changes in the ecological characteristics of the site, is essential. In Mai Po RAMSAR, the HKSAR Government is currently implementing its Conservation Strategy and Management Plan with a long-term monitoring program. Similarly in Mainland China, permanent plot studies and long-term monitoring have been carried out in Futian, Hainan Dongzhaigang, and Guangxi Shankou mangrove reserves (Lin, 1999).

11.5.4. Insect Infestation and Exotic Species Invasion

Insect infestation is another recent problem in HKSAR and Guangdong Province. Since 1990s, *Avicennia marina*, the pioneer and dominant species, has suffered from serious attacks by caterpillars of two moth species, *Pseudocatharylla duplicella* and *Oligochroa cantonella*, and *Palliphera nobilis* from April to June every year. Such attack causes massive defoliation (more than 80% of leaves were eaten) leading to a failure in the production of fruits and seeds (Wang et al., 2002). In Futian, three pest insects were recorded in 1994 with a large piece of *Av. marina* being killed, and the number increased to nine in 1999. The major insect attacking *K. candel* and *A. corniculatum* is *Amatissa* sp. although its damage is still tolerable. The deterioration of the surrounding habitats and the loss of terrestrial plants and birds are possible reasons for the pest insect outbreaks. Improving the habitat quality at the back of the mangroves in order to support a diverse group of organisms is the best biological control. In Futian Nature Reserve, plans have been suggested to restore landward vegetation and the associated animal populations.

Table 11: Concentrations of heavy metals (μg g^{-1}), polychlorinated biphenyls (PCBs μg kg^{-1}) and PAHs (polycyclic aromatic hydrocarbons μg kg^{-1}) in mangrove sediments.

Pollutant	HKSAR mean and range[a]	Polluted sites in Futian, Shenzhen mean and range[b]	Futian Shenzhen, mean and range[c]	Futian, Shenzhen mean[d]	Yingluo, Gangxi mean[d]	Jiulongjiang, Fujian mean[d]
Cd	0.67–2.26	1.6 (0–2.0)	2.96 (0.27–7.94)	0.14	0.077	0.094
Cr	3.80–129	NA	34.2 (6.8–56.7)	7.97	9.27	4.73
Cu	6.85–75.5	58.1 (7.1–127)	41.1 (15.8–308)	38.3	18.9	29.7
Mn	34–223	NA	438 (124–789)	537	583	583
Ni	4.68–30.2	NA	NA	25.0	14.6	16.9
Pb	28.9–85.4	59.8 (18.7–81.3)	35.4 (0.1–423)	28.7	10	18.3
Zn	50–263	146 (66–202)	146	114	47	111
PCBs (Tam & Yao, 2002)	3.86 (0.1–25.1)	NA	NA	NA	NA	NA
PAHs (Tam et al., 2001)	1,992 (356–11,098)	7,274 (1,218–14,466)	NA	2,196 (408–10,811)	NA	NA

NA, not available.
[a] Tam & Wong (2000b) for metals.
[b] Wang et al. (2002).
[c] Tam et al. (1995b).
[d] Lin (1999).

In addition to insects, invasion by exotic species, in particular *Mikania micrantha* — a noxious species originating from South America, has become a problem in the HKSAR and Futian mangroves. As a climber with rapid growth and reproductive rates, this weed grows over and kills the host. How to control this weed is still uncertain as the plant recovers rapidly from the remnant propagules (within 6 months) after manual cutting (Wang et al., 2002). Another exotic species causing threats to mangroves in Zhuhai, Guangdong and Mai Po, HKSAR is *Spartina anglica* (a grass) which occupies the mudflats and restricts tidal flow, thus limiting the dispersion and development of mangrove fruits, seeds and propagules.

11.5.5. Insufficient Resources and Lack of Integration Between Various Departments

Resources and manpower allocated to mangrove are often limited. Too many government departments are involved in mangrove conservation, including the Ministry of Forestry, Ministry of Agriculture, National Marine Bureau and National Environmental Protection Bureau, but without clear responsibility and coordination. Insufficient resources and lack of communication often lead to poor management, even in the core and buffer zones, and illegal intrusion and development within these zones are common in China.

To strengthen the protection and management of mangroves in China, a leading group on Mangrove Conservation and Management was formed in April 1995, and adopted the principle of integrated and coordinate mangrove conservation (Tam & Wong, 2002). In the 1990s, China stepped up its efforts to protect mangroves, with aims to conserve and restore this endangered coastal ecological system. A draft plan in 1998 was made. Local governments, in particular Fujian province has also passed and enforced regulations to protect the mangroves. In 1998, Wetland International-Asia Pacific China Program (WIAPCP), a non-profit international environmental protection organization, implemented a US$ 25,000 conservation project at Hainan Donzhaigang Mangrove Reserve, and proposed a US$ 5 million project for comprehensive protection of the Zhanjiang Mangrove Reserve. Under a project financed by the United Nations Environment Program, China will receive US$ 10 million over 10 years to protect its mangroves and their wetland habitats (website: http://www.oneworld.org/ips1/apr00/08_36_010.html). With a better understanding of mangrove characteristics, functions and problems, together with the development of proper conservation plans, and the allocation of sufficient funds, then mangroves in China would be well protected from man-made threats.

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Chapter 12

An Integrated Analysis of Sustainable Human–Water Interactions in Wetland Ecosystems of Taihu Lake Basin, East China

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Abstract. The authors in this chapter have made integrated analyses in driving forces, states and human responses in water resource, water environment, water ecosystems and water security in Taihu Lake Basin, East China. The results have shown that landuse, water resource exploitation and environmental impacts from engineering and management activities as key driving forces have caused great degradations of wetland ecosystems in the basin, so the two critical human responses to these negative changes are to use ecological engineering measures other than typical environmental engineering and life-cycle-oriented ecological management other than typical end-of-pipe management so as to maintain sustainable human–water interactions for wetland ecosystem management of the basin.

12.1. Some Background Details of the Taihu Lake Basin

The Taihu Lake basin is located in the east of China, 30°5′–32°8′N and 119°8′–120°55′E. The total area is 36,500 km^2, 0.45% of the area of China, 53% of the lake's surface belongs to Jiangsu Province, 33.4% to Zhejiang Province, 13.5% to Shanghai and 0.1% to Anhui Province (Fig. 1). There are seven medium or large cities closely linked with the lake basin, including Shanghai, Suzhou, Wuxi, Changzhou, Hangzhou, Jiaxing and Huzhou, 26 counties or county level cities, and 951 towns. The arable land area is 1.35 million ha (Sun et al., 1993; She, 1997).

This basin is in the subtropical monsoon climatic zone. The annual average temperature is 14.9–16.2°C. The temperature in July, is 27.7–28.6°C and in

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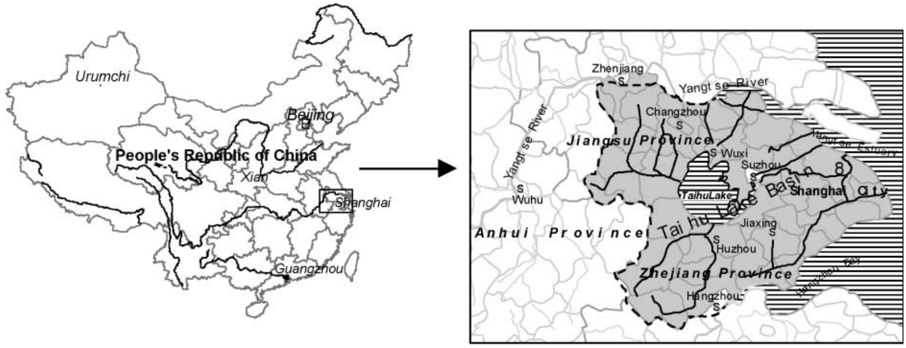

Figure 1: The geographic position of Taihu Lake Basin in China.

January, 1.7–3.9°C. The annual precipitation is 1,000–1,400 mm (Sun et al., 1993).

The topography of Taihu Lake Basin is as follows: 75% of the area, or 27,400 km^2, is plain, 25% are hilly and mountainous areas, and 17.5% water surface, including rivers and lakes. The altitude in the plain areas is 2–10 m asl, with the lowest area in the center. The central area, with an elevation of less than 3 m, encompasses the three lakes: Taihu, Yangcheng and Dianshan. In the hilly and mountainous areas, the highest point in the Tianmu Mountains, Longwangshan peak, has an altitude of 1,587 m (Table 1) (She, 1997; Nanjing Institute of Geography & Limnology, Chinese Academy of Science, 1988).

The total population in the basin was 41.51 million in 2000 with the density of 1,137 persons/km^2. There is a well-developed urbanized area, on average a city per 4,400 km^2, and a town per 38.38 km^2. The GDP per capita is RMB 256,000 in 2000 (China State Statistical Bureau, 2001).

Table 1: Topography in Taihu Lake Basin.

Altitude (m)	**Area (km^2)**	**Share of the plain (%)**
<4	9,225	33.7
(in which area of lakes)	(3,159)	(11.5)
4–5	10,740	39.2
5–6	2,280	8.3
6–7	2,750	10.0
>7	2,380	8.7
Total	27,375	100.0

12.2. Concepts and Methodology for this Research

12.2.1. Basic Concepts

Here, an essential concept of HREEES (humans, resources, environment, ecosystem, engineering, security) systems is introduced.

HREEES Systems are considered as a kind of important *human–environment interactive ecosystem (HEE)*. Generally, there are the three basic categories of the HEE, which are, respectively, environment-dominated, human-dominated and human–environment interactive subject ecosystems (shown in Fig. 2). The environment in the HEE has two components: natural and man-made. For example, a developed city is usually a human-dominated ecosystem. However, a city may be one of the three kinds of HEE ecosystems mentioned above.

The formation of the HREEES Systems can be driven mainly by human landuse, resource exploitation and environmental and ecological impacts or disturbances together with other non-human forces. From a human perspective, the two key parts in the HREEES Systems are human and environment dimensions, which are self-organized through mass, energy and information flows and therefore generate the dynamics of structure and functions in the HREEES Systems (Fig. 3).

12.2.2. The Methodology of Research

Using the concept of the HREEES Systems, we can form an overall image of a human–water ecosystem, including humans (population, cultural and social progresses), water resources, water environment, water ecosystems and water security, which forms various interactions driven mainly by landuse, resource exploitation, environmental and ecosystems impacts (L.R.E.). The methodological

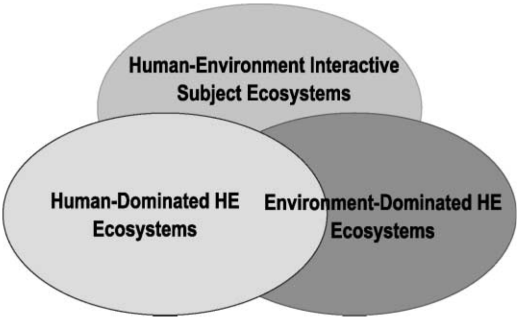

Figure 2: The three basic categories of human–environment ecosystems (HEE).

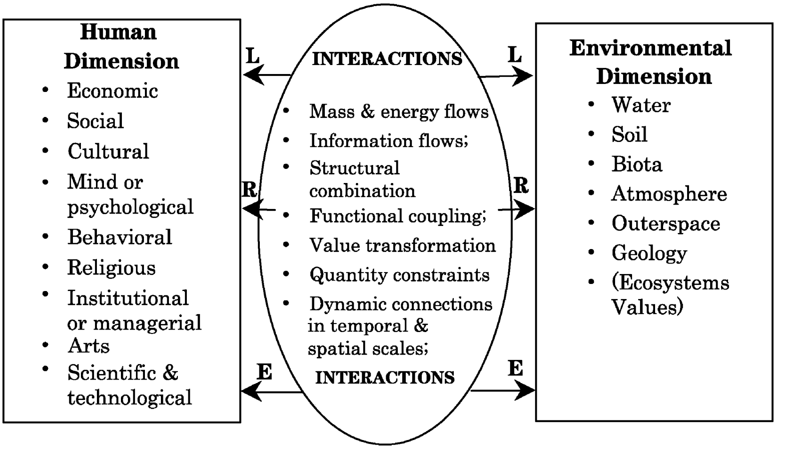

Figure 3: An integration interaction for the formation of the HREEES Systems by landuse (L), resources exploitations (R) and environmental and ecological impacts (E).

framework of driving forces–states–responses can be used to describe the complicated changes in the human–water wetland HREEES ecosystem in the Taihu Lake Basin at different rates and on different scales according to time and location, and the sustainable measures can also be explored in terms of the security of the wetland HREEES Ecosystems (Fig. 4).

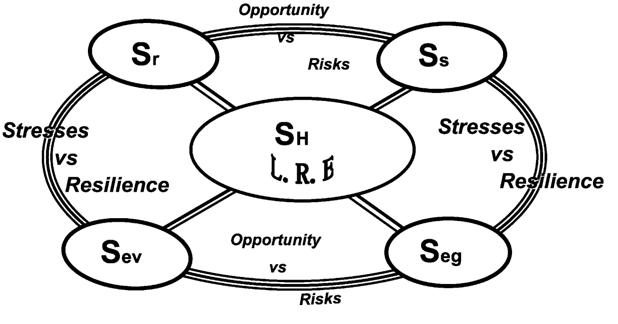

Figure 4: Research methodology chart of human–water HREEES Ecosystems. Notes: (1) **SH**: human developmental security such as population growth, cultural and social progresses; (2) **L.R.E.**: driving forces from human's landuse, resources exploitations, environment and ecosystems impacts; (3) **Sev**: water environmental security; (4) **Seg**: engineering security for water; (5) **Sr**: water resources security; (6) **Ss**: aquatic ecosystems security; (7) "▬▬ ═══" temporal, spatial pathways of driving forces state responses (DSR) for human water interactions; (8) "▬▬" security circles.

12.3. Hydrology, Water Resources and Water Disasters in the Taihu Lake Basin

12.3.1. Hydrology in the Taihu Lake Basin

The Hydrological System in the Taihu Lake Basin. The sources of the water system in the Taihu Lake Basin include the Jingxi River and the Tiaoxi River. The former originates from Tao Lake, the Ge Lake water system and the piedmont areas of Yixing and Liyang Mountain and Mao Mountain. The latter originates from Tianmu Mountain. The main water outlets of Taihu Lake have four routes. The first goes through the Shadunkou and Wangyuhe Rivers into the Yangtze River at Huazhuang of Changshu city. The second runs through Xukou, then through Loujiang River into the Yangtze River by Liuhe in Taichang county. The third goes through Kuajingkou, then into the Wusong River to the Huangpujiang River. The last runs through Taipu Gate and some small lakes, to the Dapuhe River and the Huangpujiang River, then into the Yangtze River.

The Taihu Lake Basin consists of 219 river courses. The total length of these rivers is 120,000 km, and the main watercourses have been extended to about 1,200 km. Major rivers in the Taihu Lake Basin include the Grant Canal, and the Huangpujiang, Tiaoxi, Nanxi, Taipu, and Wangyu Rivers. The topography in the basin is flat and the rivers form a fan-shaped drainage system at the lower reaches. There are 189 lakes with an area greater than 0.5 km^2, and 9 lakes with an area greater than 10 km^2. The overall area of the lakes in the basin is 3,159 km^2, and that of the reservoirs is 73 km^2 (Table 2) (Wang et al., 1989; Sun et al., 1993).

Hydrological Zones of the Taihu Lake Water Ecosystem. According to the hydrological characteristics of the basin, we can divide the basin into seven hydrological subregions as follows (Sun et al., 1993; Han & Mao, 1995; She, 1997).

(1) West Taihu Lake Region This region includes Zhenjiang, Changzhou, Wujin, Danyang, Jintan, Liyang, and Yixing city. The rivers, connecting and crisscrossing with the Yangtze River, Canal and Taihu Lake, mainly come from Mao Mountain and Yixing-Liyang Mountain. Most of the water in these rivers flows into Taihu Lake. There are some reservoirs in this region, which, together with Taohu Lake and Gehu Lake, can buffer some floodwater, and adjust runoffs flowing into the Nanxi River.

(2) East Taihu Lake Region There are many cities and counties located in this region, including Changshu, Suzhou, Wuxian, Wujiang, Kunshan, Taicang and so on. This region can be further divided into two small areas by the Shanghai-Nanjing railway line: to the north of the line is the area of Yangchenghu Lake, and to the south is the area of Dianshanhu Lake and Maohu Lake. The area of

Table 2: Lakes in Taihu Lake Basin.

Region	Lake area (km^2)									
	>10 km^2		10–5		5–1		1–0.5		Total	
	No.	Area	No.	Area	No.	Area	No.	Area	No.	Area
YCDM	6	264.34	8	52.10	34	72.62	22	15.20	70	404.26
PHJH			4	29.30	44	78.95	43	20.10	91	128.35
Puxi					1	1.63	3	1.52	4	3.15
Pudong										
Chengxiyu			1	5.29	2	2.79	1	0.65	4	8.73
Huxi	2	235.85	3	21.57	5	13.17	3	2.27	13	272.86
Zhexi							6	3.52	6	3.52
Taihu	1	2,338.1							1	2,338.1
Total	9	2,839.29	16	108.26	86	169.19	38	43.26	189	3,158.97

YCDM: Yangcheng Lake, Cheng Lake, Dianshan Lake and Mao Lake; PHJH: the plain of Hangzhou, Jiaxing and Huzhou city.

Dianshanhu Lake and Maohu Lake is the secondary area for buffering and storing water coming from Taihu Lake.

(3) Chengxiyu Region This area lies between the Chengxi Canal and the east bank of the Wangyuhe River. In the northern part of this region is the Yangtze River and in the south is Taihu Lake. It includes the cities of Wuxi, Jiangyin, Changshu and Zhangjiagang. Floodwater in this region will mainly flow into the Yangtze River. Some water flows into the Grand Canal, and exchanges water with Yangchenghu Lake, Dianshanhu Lake and Maohu Lake; other flows water into the Wangyuhe River, then into the Yangtze River. In the flood season, this region can buffer some water from Taihu Lake and in the dry season, this region can pump water from the Yangtze River for irrigation and navigation.

(4) West Zhejing Region This region includes Changxin, Anji and Lin'an in Zhejiang province. The Tiaoxi River is the largest in this area. Some of the water from the mountainous areas is absorbed by the reservoirs, and of the other runoffs flow directly into Taihu Lake or lakes on Hangzhou-Jiaxing-Huzhou Plain via the East Tiaoxi River.

(5) East Hangzhou-Jiaxing-Huzhou Plain Region This region includes the area from the south of the Taipu River to the east of the Tiaoxi River. To the east of the region are the Huangpujiang and Zhangjinghe Rivers. To the southeast is the Qiantangjiang River and Hangzhou Bay. The region consists of the rich Hangzhou-Jiaxing-Huzhou plain and the southern area of the Taipuhe River in

Zhejiang province. The southeast of this area is slightly higher and the center is low. There are many rivers and lakes crisscrossing the area, into which the floodwater from the Tiaoxi River is often discharged.

(6) Huangpujiang River Valley Region This region includes Dianshanhu Lake, as well as cities and counties such as Shanghai, Jiading, Baoshan, Pudong, Shuangsha, Nanhui, Fengxian, Songjiang, Jinshan and Qingpu. It is close to the sea and located in the lower reaches of the Taihu Lake Basin. The water comes from Taihu Lake and tide. Tidal water is plentiful and evenly distributed between seasons and years. However, the tide brings about water disasters such as salt tides and waterlogging, so this region is affected by tide and the threat of flood. Its capacity to buffer floodwater is low. Another problem in this region is serious water pollution. In the low rainfall season, the water quality is affected by the inflowing salt tide.

(7) The Central Area of Taihu Lake The length of the Taihu Lake from north to south is 68.5 km, and its average width from east to west is 34 km, the maximum width being 56 km. The average depth of water is 1.9 m and the deepest is 2.9 m. The water surface area is 2,427.8 km^2, which is 139.9 km^2 less than in the 1960s. Deducting the area of several islands, the actual water surface is 2,338.1 km^2. The lake volume is 4.43 billion m^3.

12.3.2. Water Resources in Taihu Lake Basin

Available Quantity of Water Resource. The annual amount of available water can be estimated according to the following formula (Sun et al., 1993; She, 1997):

$$W_t = W_r + W_g - W_{rp}$$

In the formula, W_t is the annual volume of available water (10^8 m^3), W_r is the runoff volume from precipitation (10^8 m^3), W_g is the volume of groundwater supply (10^8 m^3). W_{rp} is the repeated amount which is the difference between the annual underground water replenishment and the annual river runoffs (10^8 m^3).

(1) Amount of precipitation The precipitation in the basin is 1,000–1,400 mm. In the Hangzhou-Jiaxing-Huzhou plain, the precipitation is higher than in the northern region. The precipitation in the Huangpujiang Watershed system is higher than in the western part of the basin. The southern mountainous areas get more precipitation than the northern plain areas. The precipitation varies greatly throughout the year and from year to year due to the influence of the monsoon. The precipitation in the more rainy years is 1–1.5 times higher than that in the less rainy years. The relative variability is 15–30%. The peak value for precipitation appears in the period from June to August, making up about 35–40% of the annual precipitation; the lowest value is from December to February at only 11–14%.

(2) Amount of runoff flow The runoff coefficient in the Taihu Lake Basin is 0.26–0.4. The coefficients in the mountainous area and the Hangzhou-Jiaxing-Huzhou plain are higher than those in the northern plain areas. The average annual runoff in the basin is 14.17 billion m^3 (Table 3), of which 5.54 billion m^3 is in Jiangsu Province, 7.07 billion m^3 in Zhejiang Province and 1.56 billion m^3 is in Shanghai. The area of Jiangsu Province is about 8,000 km^2 greater than that of Zhejiang Province; however, the runoff in Jiangsu Province is 1.53 billion m^3 less than the runoff in Zhejiang Province. In Zhejiang Province, the average volume of water generated is 582,000 m^3/km^2 in Shanghai 305,000 m^3/km^2 but in Jiangsu Province, only 274,000 m^3/km^2.

(3) Volume of groundwater The groundwater table in the basin is high. Its recharge mainly depends on the precipitation. The total amount of shallow groundwater in Taihu Lake Basin is 5.52 billion m^3 the amount of replenishment is 2.32 billion m^3. In the eastern areas of the Huangpujiang Watershed, the recharge volume is the greatest, 0.61 billion m^3; then, in the west of the Taihu Lake Basin, it is 0.59 billion m^3. In these areas, the groundwater resource is relatively plentiful.

(4) Local water resources The local water resource is the sum of the surface water and the groundwater excluding the river outflow, or the sum of the local outflow and the replenishment of the subsurface water. In the Taihu Lake Basin, the annual local water resource is 16.5 billion m^3, 41% of which (6.79 billion m^3) is in Jiangsu Province, 45.7% (7.55 billion m^3) in Zhejiang Province and 11.3% (2.16 billion m^3) in Shanghai.

(5) Volume of water inflow The upper reaches of the Taihu Lake Basin is a relatively closed watershed and there is no inflow; however, it is between the Yangtze River and the Qiantangjiang River and close to the East China Sea, so water from rivers and tides can be pumped into this basin. Therefore, the water resource from outside is plentiful. The total water inflow in the Basin is 71.69 billion m^3 ($P = 50\%$) (Table 4). In the part of the basin within Jiangsu Province, the water resource is mainly from the Yangtze River, and the inflow is up to 14.54 billion m^3 ($P = 50\%$), whereas in Shanghai, the inflow is tidewater from the Huangpujiang River and the total inflow is 55.48 billion m^3 ($P = 50\%$).

(6) Total volume of the water resource The total volume of the water resource in the Taihu Lake Basin under different rates of guarantee (P) 87.66 billion m^3 ($P = 50\%$), 85.5 billion m^3 ($P = 75\%$) and 78.87 billion m^3 ($P = 95\%$), respectively. In drier years, the water resource is 9.79 billion m^3 ($P = 50\%$) and the sum of the water resource in Jiangsu Province and Zhejiangjiang Province is 29.52 billion m^3 ($P = 50\%$) (Table 5) (Sun et al., 1993).

Volume of the Water Resource Per Capita in the Taihu Lake Basin. In the Taihu Lake Basin, the average annual volume of local water resource per capita is

Table 3: Local water resource in Taihu Lake Basin.

	Area (km^2)	**Annual precipitation (mm)**	**Annual runoff (mm)**	**Local runoff (10^8 m^3)**	**Ground-water (10^8 m^3)**	**River runoff (10^8 m^3)**	**Local water resource (10^8 m^3)**
West of the Basin	8,880	1,079	319	28.34	10.04	4.17	34.21
East of the Basin	5,307	1,044	285	15.10	7.47	3.67	18.90
Surface of Taihu	2,338	1,069	69	1.59			1.59
Plain along Huangpujiang river and the sea	5,110	1,141	582	15.59	11.09	5.05	21.63
Cheng-Xi-Yu areas	3,705	1,036	281	10.42	5.37	2.61	13.18
Shaoxi mountainous areas	5,797	1,333	305	40.20	11.41	11.41	40.20
Hangzhou-Jiaxing-Huzhou plain	6,357	1,333	582	30.50	9.82	5.04	35.28
Total				141.74	55.20	32.05	164.99

Table 4: Inflow amount in Taihu Lake Basin under different rates of guarantee (P).

	$P = 50\%$	$P = 75\%$	$P = 95\%$
West of the Basin (m^3)	54.19×10^8	61.21×10^8	70.14×10^8
East of the Basin (m^3)	60.84×10^8	53.18×10^8	70.36×10^8
Cheng-Xi-Yu area (m^3)	30.33×10^8	32.00×10^8	43.80×10^8
Plain along Huangpujiang river and the sea (m^3)	554.8×10^8	554.8×10^8	500.7×10^8
Shaoxi mountainous areas (m^3)		0.6×10^8	2.59×10^8
Hangzhou-Jiaxing-Huzhou plain (m^3)	16.72×10^8	24.62×10^8	24.21×10^8
Total (m^3)	716.88×10^8	726.41×10^8	711.8×10^8

Note: P denotes the rate of guarantee.

861 m^3, which is less than 1/5 of that in the country as a whole. The water resource per ha in the Taihu Lake Basin is less than 2/5 of that in the country as a whole. Furthermore, the water resource per capita is highest in Zhejiang (1,751 m^3), is lowest in Shanghai (228 m^3), and 469 m^3 per capita in Jiangsu Province. If the

Table 5: Amount of water resource in Taihu Lake Basin under different rates of guarantee.

	P (%)	**River runoff (10^8 m^3)**	**inflow (10^8 m^3)**	**total (10^8 m^3)**	**Replenishment from subsurface water (10^8 m^3)**	**Total volume of water resource (10^8 m^3)**
Jiangsu	50	54.64	145.36	200.01	12.43	212.43
	75	33.97	146.39	180.36	12.43	192.79
	95	7.98	184.3	192.28	12.43	204.56
Shanghai	50	20.41	554.8	575.21	6.04	581.25
	75	19.35	554.8	573.35	6.04	579.39
	95	5.88	500.7	506.58	6.04	512.62
Zhejiang	50	61.31	16.72	78.03	4.78	82.81
	75	39.41	25.22	64.63	4.78	69.41
	95	30.03	26.80	56.63	4.78	61.41
Total	50	136.18	716.88	853.34	23.25	876.59
	75	92.73	726.41	831.74	23.25	854.99
	95	43.89	711.8	755.49	23.25	778.74

Note: P denotes the rate of guarantee.

total water resources, including local water and inflow water resources, are counted together, the water resource per capita is 2,666 m^3 ($P = 50\%$). It is highest in Shanghai, 5,389 m^3, lowest in Jiangsu Province 1,807 m^3 and 2,334 m^3 in Zhejiang Province.

Water Supply and Demand

(1) Volume of water available The volume of water resource available in the basin is 35.16 billion m^3 ($P = 50\%$) and 36.73 billion m^3 ($P = 90\%$) (Table 6). The rate of water use is about 40–47%. In Jiangsu Province, the available water resource is 23.19 billion m^3, which exceeds the total amount of water resource due to the large-scale use of water from the Yangtze River and some reuse of water or wastewater. The ratio of reused water is 72.4%. In Shanghai, the water resource is abundant, but part is tidewater so that there are limitations on its use. It was expected that the available water resource could be increased to 34.47 billion m^3 ($P = 50\%$), 35.44 billion m^3 ($P = 75\%$) and 36.73 billion m^3 ($P = 95\%$) in 2000 (Table 6) (Nanjing Institute of Geography & Limnology, Chinese Academy of Science, 1988; Sun et al., 1993; Sun, 1995; She, 1997).

(2) Water demand The demand for water resource by different sectors including industry, agriculture and domestic uses under the different rates of guarantee is listed in Table 7. The average annual demand for water is 27.58 billion m^3 ($P = 50\%$), 28.86 billion m^3 ($P = 75\%$) and 32.03 billion m^3 ($P = 95\%$), of which the agriculture water demand accounts for 67.2% (18.54 billion m^3 ($P = 50\%$)). In addition, agriculture water use in Jiangsu province the largest consumption, accounting for 12.48 billion m^3, which is 67.18% of the total in the basin. The total amount of industrial water used is around 8 billion m^3, 4.45 billion m^3 of which is used in Shanghai. The domestic water consumption in the basin is relatively small, 0.54 billion m^3, representing only 2.0% of the total demand. The domestic water consumption in Shanghai is the highest, at 70.8% of the total

Table 6: Available water resource in Taihu Lake Basin.

Region	**50% rate of guarantee**		**95% rate of guarantee**	
	Total water resource (10^8 m^3)	**Available water resource (10^8 m^3)**	**Total water resource (10^8 m^3)**	**Available water resource (10^8 m^3)**
Jiangsu	212.43	231.89	204.71	231.89
Zhejiang	82.81	34.76	61.41	42.42
Shanghai	581.25	80.35	512.62	92.99
Total	875.93	351.63	778.74	367.30

Table 7: The demand for water in the Taihu Lake Basin with different guarantee rates.

Area	Water demand by agriculture (× 10^8 m^3)	Water demand by industry (× 10^8 m^3)	Water demand by cities (× 10^8 m^3)	Water demand by rural areas (× 10^8 m^3)	Total (× 10^8 m^3)
P = 50%					
West of the lake	51.37	11.52	0.23	1.12	64.25
East of the lake	48.81	3.86	0.35	0.85	53.86
Zheng-Xi-Yu area	24.66	16.72	0.47	0.77	42.61
Plain areas along Huangpujiang river and the sea	30.60	44.49	3.84	1.43	80.36
Shaoxi Brook	7.14	0.22	0.04	0.27	7.67
Hangzhou-Jiaxing-huzhou plain	23.26	2.47	0.49	0.87	27.09
Total	185.84	79.28	5.42	5.31	275.84
P = 75%					
West of the lake	54.29	11.52	0.23	1.12	64.25
East of the lake	45.28	3.86	0.35	0.85	50.35
Zheng-Xi-Yu area	26.68	16.72	0.47	0.77	44.63
Plain areas along Huangpujiang river and the sea	37.40	44.49	3.84	1.43	87.15
Shaoxi Brook	8.86	0.22	0.04	0.27	10.38
Hangzhou-Jiaxing-huzhou plain	28.00	2.47	0.49	0.87	31.83
Total	198.51				288.59
P = 95%					
West of the lake	63.21	11.52	0.23	1.12	76.09
East of the lake	54.20	3.86	0.35	0.85	59.26
Zheng-Xi-Yu area	30.54	16.72	0.47	0.77	48.49
Plain along Huangpujiang river and the sea	43.24	44.49	3.84	1.43	92.99
Shaoxi Brook	9.85	0.22	0.04	0.27	10.38
Hangzhou-Jiaxing-huzhou plain	29.26	2.47	0.49	0.87	33.09
Total					320.3

Note: *P* denotes the rate of guarantee.

in the basin (Nanjing Institute of Geography & Limnology, Chinese Academy of Science, 1988; Sun et al., 1993; She, 1997).

(3) The equilibrium between water supply and demand The water resources in the Taihu Lake Basin can meet demand under different rates of guarantee (Table 8) (Nanjing Institute of Geography & Limnology, Chinese Academy of Science, 1988): there is a 7–14 billion m^3 surplus of water. However, in recent years, industrial and agricultural production has increased rapidly. In typical years, the shortage of water is 2 billion m^3, but in a very dry year it can be more; for example, in 1978 the shortage was 12 billion m^3.

The total volume of water demand in the Basin is 57 billion m^3 if the rate of guarantee is 95% during the peak period of water consumption (1971 was a typical year), of which the demand for water in agriculture is 20.2 billion m^3, for industry 27 billion m^3, for domestic uses 2 billion m^3 and for the environment 6.8 billion m^3. However, the average annual water resource in total is 16.5 billion m^3. In 1971, a dry year, it was 8 billion m^3 ($P = 94\%$), in 1978, a seriously dry year, only 1.6 billion m^3 ($P = 98\%$) (Sun et al., 1993; Sun and Mao, 1994; Sun, 1995).

12.3.3. Water Disasters in the Taihu Lake Basin

The Taihu Lake Basin has very frequent flooding and drought (including coastal storm tides). In the last century, there were six individual years in which flooding occurred. In 1954, the Taihu Lake Basin was hit by a large flood with 21.2 billion m^3 of flooding; 523,300 ha of land were affected by the disaster, of which 248,670 ha suffered greatly: direct economic losses were 1 billion RMB yuan. It appears that the frequency of large floods is about every 5–10 years (Sun et al., 1993).

There were six individual years in which drought occurred within the 20th century. For example, a serious drought in 1934 caused a water shortage of 1.3 billion m^3, affecting 68,700 ha of land (Sun et al., 1993).

Although the Taihu water system has been improved, it still faces a severe risk of flooding. Recently there have been some changes as follow (Wang, 1994):

(1) In the 1950s, there was 900 mm of rain in the flooding periods and the water level in Taihu Lake was only 4.0 m asl (above sea level); nowadays, there is 300–400 mm rain in the flooding periods but the water level in Taihu Lake still goes beyond 3.5 m (the warning water level in Taihu Lake is 3.5 m asl).
(2) In the early years of the 1990s, high water level appeared three times, the highest water level in Taihu Lake was 4.79 m in 1991, 4.51 m in 1993 and 4.32 m in 1995.

Table 8: Water equilibrium under different rates of guarantee in the Taihu Lake Basin (in 10^8 m^3).

Area	Water resources	Available water resource	Total demand for water	Surplus or shortage of water	
				Surplus	Shortage
P = 50%					
West of the lake	90.29	98.17	64.25	37.8	3.89
East of the lake	76.38	73.85	53.86	29.18	9.21
Zheng-Xi-Yu areas	45.76	64.50	42.61	21.88	
plain areas along Huangpujian and the sea	581.25	80.35	80.36		8
Shaoxi brook	31.39	7.67	7.67	0	0
Hangzhou-Jiaxing-huzhou plain areas	51.42	27.09	27.09	0	0
Total	876.49	352.44	275.84	140.16	21.39
P = 75%					
West of lake	81.58	90.26	67.19	23.11	0
East of lake	70.39	79.94	50.35	29.59	0
Zheng-Xi-Yu area	40.82	61.63	44.63	17.00	0
Plain areas along Huangpujian and the sea	579.39	87.15	87.15	49.3	0
Shaoxi brook	20.95	9.39	9.39	0	0
Hangzhou-Jiaxing-huzhou plain area	48.46	31.83	31.83	0	0
Total	841.59	360.2	290.54	119.02	0
P = 95%					
West of lake	80.93	93.82	76.09	20.80	3.06
East of lake	76.26	74.75	59.26	15.91	0.42
Zheng-Xi-Yu area	47.52	63.32	48.49	18.37	3.54
Plain areas along Huangpujian and the sea	512.62	92.99	92.99		
Shaoxi brook	18.05	9.33	10.38	0	1.05
Hangzhou-Jiaxing-huzhou plain areas	43.36	33.09	33.09	0	0
Total	778.74	367.30	320.30		

Note: *P* denotes the rate of guarantee.

(3) In 1991, there was a rainfall with a frequency of every 20 years, and a high water level with a frequency of above 50 years had appeared.
(4) In the flooding period, the maximum daily water level rise increases year by year. In 1954, the maximum daily water level rise was only 9 cm/day, but in 1991, it became 13 cm/day, and reached 22 cm/day in 1995.

12.4. Water Quality Changes in the Taihu Lake Basin

12.4.1. Pollution Sources from Urbanization and Industrialization

Pollution sources include wastewater discharge from various urban activities. According to the statistical data in 1994, the total discharge of wastewater into the basin was 3,200 million t per annum (now 3,500 million); chemical oxygen demand (COD_{Cr}), 282,404 t; total nitrogen (TN), 79,522 t; total phosphorus (TP), 5,660 t (Table 9). The discharge of wastewater into the basin accounts for 1/10 of national discharges and 1/3 of that into the Yangtze River Valley. Wastewater discharge per unit area reached 109,000 t per annum. 80% of the wastewater was untreated, or treated but not meeting the State Standard for Wastewater (Cai, 1995).

The chemical fertilizer applied in the basin is 2–3 million t annually (1,500–3,000 kg per ha), and pesticide, 750,000–1,200,000 t (42 kg per ha). The efficiency of use of chemical fertilizer is less than 50%. Nearly half the chemical fertilizer flows into waters.

The ranking of contribution (CR) for different pollution sources of COD_{Cr} to the water pollution of rivers and lakes from high to low is industry, domestic, husbandry, precipitation and dustfall; the ranking of CR for TP into rivers and lakes from high to low is domestic, husbandry, fisheries, plantation, precipitation and dustfall; and the ranking of CR for TN into rivers and lakes from high to low is domestic, plantation, husbandry, fisheries, precipitation and dustfall. In addition, the COD discharged into the Yangtze River from the Huangpujiang River per year is 50.8 thousand t, Hg, 3.0 t, Cu, 238.9 t, Zn, 1,300 t, Al, 249.0 t, Cd, 1.80 t, oils, 0.483 t (Cai, 1995).

In about 48 small cities and towns in the Taihu Lake Basin in 1987 (Cai, 1995), the total wastewater discharge was about 371 million t/yr, of which industrial wastewater accounted for nearly 90%. The number of towns whose total wastewater exceeded 10 million t/yr is 12. There were 14 towns whose total wastewater was 5–10 million t/yr. Those whose wastewater was less than 5 million t increased to 22. In 1985, the wastewater discharged by the major industrial pollution sources in 12 cities and counties was 209.7 million t, of which the

Table 9: Pollution sources in Taihu Lake Basin (1994).

Pollution sources	**Wastewater amount (10^4 t/yr)**	**COD_{Cr} (t/yr)**	**CR of COD_{Cr} (%)**	**TN (t/yr)**	**CR of TN (%)**	**TP (t/yr)**	**CR of TP (%)**
Industrial wastewater in major trades	53,901	111,061	39.3	12,544	15.8	591	10.4
Domestic sewage	32,290	119,029	42.2	19,948	25.1	3,394	60.0
Farmland loss and erosion[a]	12,8373			18,355	23	164	2.9
Scattered inhabitants[a]	15,671	11,377	4.0	1,896	2.4	433	7.65
Husbandry[a]	1,203	16,761	5.9	9,591	12.0	255	4.5
Sum[a]	14,5247	28,138	10.0	29,842		852	
Fisheries	83,774			13,195	16.6	533	9.4
Domestic pollution of lakes	216	417	0.15	21	0.03	3	0.05
Precipitation on lake surface	3,341	23,595	8.4	2,760	3.4	60	1.06
Dustfall				421	0.5	33	0.58
Ships		164	0.06	22	0.03	2	0.03
Water loss and soil erosion				800	1	192	3.39
Total	318,769	282,404		79,552		5,660	

[a] Non-point sources in rural areas.

treated wastewater was only 11.7%, while those up to the State emission standard after treatment only occupied 1.8%.

Pollution discharge every 10,000 yuan of output value in the Taihu Lake Basin are, respectively: papermaking, 1,673 t; chemical industry, 1,358 t; food, 324 t; tannery, 248 t; electroplating, 247 t; and textile and printing and dyeing, 174 t.

12.4.2. Present State of Water Quality in the Taihu Lake Basin

In the basin, according to the State Water Quality Standard, some reservoirs in Tianmushan areas in Zhejiang province have Class I water quality, and east and west Tiaoxi River has relatively good water quality. The water quality of the Taipuhe and Wangyuhe Rivers (the two main river courses) and the rivers surrounding Taihu Lake is very bad.

The Water Quality of Different Sections of River Courses. In the basin, the river sections whose water quality is worse than Class V for all periods 181.9 km, which is 18.84% of the total assessed length of rivers. The sections whose water quality was above Class V in all three periods were 65.35% of the total evaluated length of the sections.

The length of contaminated river whose water quality is worse than Class IV in the flood season, non-flood season, and in one full year is 786, 883 and 841 km, respectively, accounting for 68, 77 and 73 of the total assessed sections. The pollution is mainly caused by COD, NH_3-N and other organic pollutants (Table 10) (Cai, 1995).

Water Quality of the Lakes. The water quality changes in Taihu Lake can be divided into three stages. In the early 1980s, the water quality of Taihu Lake was predominantly Class II, in the late 1980s, the water quality underwent a transition period from Class II to Class III and, in the middle 1990s, the water quality in Taihu Lake was mainly Class III.

The water quality of rivers into or out of Taihu Lake deteriorated day by day. By 1995, over half of the rivers had Class V or worse in 1995. The main pollutants were TP, TN and COD_{Mn}. In the flood season, the water quality was better than in the non-flood season (Table 11).

Overall, from the early 1980s up to now, the water quality in Taihu Lake degraded by one class, from Class II to Class III. Most areas of Taihu Lake reached eutrophic level. Wulihu Lake and Meilianghu Lake have reached the over-eutrophication level.

The area of Dianshanhu Lake is 63.73 km^2. In 1992, the area of Class III in the whole year was 33.20 km^2 (52.1% of the total area) and that of Class IV,

Table 10: The water quality for different sections of some rivers in Taihu Lake Basin in 1992.

River name	Assessed river length (km)	Class II		Class III		Class IV		Class V		Worse than Class V	
		Length	Ratio (%)	Length	Ratio (%)	Length	Ratio (%)	Length	Ratio (%)	Length	Ratio (%)
Canal in the south of Yangtze	299.5			51	17	81	27.1	42	14.0	125.5	41.9
Taipu River	48			10	21	13	27.0	25	52.0		
Huangpujiang River	81.3					56.4	69.4	24.9	30.6		
Suzhouhe River	45.8							11.6	25.3	34.2	74.7
East Tiaoxi River	152.5	28.5	18.7	59.0	38.7	65.0	42.6				
West Tiaoxi River	197.5	47.0	23.8	51.5	26.1	68.5	34.7	30.5	15.4		
Wangyuhe River	3					3	100				
Liuhe river	10					10	100				
Licaohe River in Danjin	10					10	100				

Note: this assessment was conducted during three different periods: the flood season, the non-flood season and for the total year (Cai, 1995).

Table 11: The state of the water quality in Taihu Lake Basin, 1992.

Rivers and lakes	**Length of river courses (km)**	**Length of assessed river courses (km) (*A*)**	**Length of polluted river courses (km) (*B*)**	**B/A (%)**	**Main pollutants**
Canal in the south of Yangtze River	312	299.5	248.5	83	Non-ionized ammonia, COD, BOD_5
Huangpujiang River	113	81.3	81.3	100	DO, non-ionized ammonia, volatile phenol, COD_{Mn}
Taipuhe River	57.6	48	25	52	Volatile phenol
Wangyuhe River	60	3	3	100	NH_3-N, DO, COD_{Mn}
Taihu Lake				21.6% is moderate eutrophication; and 78.4% over eutrophication	TN,TP

Note: Length of polluted river courses refers to Class IV or worse.

30.53 km^2 (47.9%). The perennial mean concentration of TP was 0.103 mg/l and the mean concentration of TN was 2.217 mg/l (Shi & Liu, 1989; Song et al., 1992).

Yangchenghu Lake (119.0 km^2) is, on average, Class III the whole year. The perennial average TP was 0.056 mg/l, TN, 3.02 mg/l. All the areas had moderate levels of eutrophication for TP and high levels of eutrophication for TN.

The area of Gehu Lake is 187.0 km^2. The area with Class II was 46.7 km^2 (occupying 25.0%) and that with Class IV, 140.3 km^2 (75.0%). Perennially, 50% of the lake exhibited moderate levels of eutrophication.

The total area of the four lakes mentioned above is 2,707.73 km^2. The area of Class II during the whole year was 1,631.70 km^2 (60.3% of the total area) that of Class III, 411.21 km^2 (15.20%) and that of Class IV, 663.83 km^2 (24.5%).

12.5. Driving Forces for Changes in the Taihu Lake Wetlands Ecosystems

12.5.1. Driving Forces for Water Resources Changes

The major factors resulting in changes in water resources are as follow (Sun & Mao, 1994; Han & Mao, 1995; Sun, 1995; Wang & Cheng, 1996; She, 1997):

Small runoff and uneven distribution Precipitation is the main source of runoff in Taihu Lake Basin. The variation of the precipitation in any one year and between years is very great due to the annual variation in intensity and frequency of the monsoon. In summer, precipitation is twice as large as in winter. In general, the maximum precipitation occurs in the plum rain period (a kind of typical raining season in the lower reach of Yangtze River from June to July) and the typhoon period (from August to September). The runoff during these four months is the largest, about 50–70% of the total runoff each year. The runoff modulus in the basin is only 378,000 m^3/km^2/yr, its total amount is less than 5.31 billion m^3/km^2/yr in the Yangtze River Watershed.

No effective engineering measures to control lake water In the basin, there are many other lakes besides Taihu Lake. There is no floodgate at most of the river mouths, and many riverways connecting to the lakes are not under control. Every year, more than a billion m^3 of unused water runs freely into the East China Sea.

Large scale land reclamation causes many lake wetlands and rivers to disappear There are many low-lands and lake shoals below 4 m (Wusong sea level), all reclaimed as dyked land. According to an investigation in 1990, there were about 7,000 reclamation sites, of which 500 (total area of 500 km^2) were dyked (Sun et al., 1993; Han & Mao, 1995; Sun, 1995).

Water quality of lakes and rivers decreased, reducing the availability of water resource Water shortages due to water pollution in the Taihu Lake Basin have occurred frequently in recent years. The eutrophication in Taihu Lake, Yangchenghu Lake and many others becomes progressively more serious. In the period from July to August in 1994, the weather was very dry, the water was polluted in the Huangpujiang River and the pollution extended upstream. The length of the polluted zone was 30–40 km. The water-withdrawal pipes of the waterworks along the Huangpujiang River were surrounded with polluted water (Cai, 1995; Wang & Cheng, 1996).

Water quality at the lower reaches affected by tidewater The mouth of the Yangtze River has a relatively strong tide. The Huangpujiang River is one of the main rivers flowing into the mouth of the Yangtze River, which experiences tides twice a day. In general, the tide entering Shanghai is usable freshwater. The average annual amount of tidewater is 44.25 billion m^3, but from November to April next year, salted tidewater will enter the mouth of the Yangtze River, and water in Huangpujiang River easily degrades. In brief, although tidewater makes up for the shortage of surface runoff, it causes salted tidewater to flow upstream and reduces the water quality.

Water use per product is high, bringing about an increase in water demand Compared with developed countries, the water demand per unit product and per unit output value in China is evidently high. The use of water per ton of steel production is 20–40 t, but in developed countries, only 3–5 t; the use of water per ton of paper production in China, is 200–500 t and in developed countries, around 100 t; the use of water per RMB 10,000 of output value in China is close to 400 t, but in developed countries, only about 40 t. In China, the rate of water recycling is about 25%, whereas in developed countries, it was 60% even in the late 1960s (Sun, 1995).

12.5.2. Driving Forces for Water Security Changes

There are several driving factors for the changes in flooding and drought in the Taihu Lake Basin as follow (Sun et al., 1993; Wang, 1994).

(1) There is an uneven distribution of annual and multiple-year rainfalls in the Taihu Lake Basin. The spatial distribution of annual rainfall assumes a pattern with high levels in the southwest and low levels in the northeast. The annual distribution of rainfall in the Taihu Lake Basin has two peaks, in June and September; the lowest value occurs in December and January. The rainfall in June and July largely comes from intermittent drizzle. In Shanghai, the rainfall in Autumn, caused by typhoon weather, accounts for 30 and 38% of total annual rainfall.

The discrete coefficient (C_v value) of monthly rainfall at the Xishan hydrological station (Table 12) shows that the multiple-year rainfall is much

Table 12: The discrete coefficient of monthly rainfall in Xishan (C_v).

Month	**1**	**2**	**3**	**4**	**5**	**6**	
C_v	0.6958	0.5718	0.5062	0.4173	0.4688	0.4541	
Month	**7**	**8**	**9**	**10**	**11**	**12**	**Year**
C_v	0.5287	0.7303	0.7185	0.8955	0.7350	0.8100	0.1904

larger than the annual rainfall (Sun et al., 1993). That is to say, since Autumn, the C_v value obviously rises, indicating an increase in multiple-year rainfall. The period from Autumn to October is one of high water demand for crop growth, but the monthly rainfall decreases gradually; furthermore, Autumn is the high evaporation period, thus the increase in multiple-year rainfall easily caused water disasters in the period. Although the C_v value in June and July was smaller than that in other months, the monthly rainfall was very large, so the uneven rainfall readily caused flooding and waterlogging disasters.

(2) The inflow is larger than the outflow in Taihu Lake: in the 1991 flooding period, the water level in the lake was high, and the inflow from the upper reach of the west Taihu Lake was large and fast, making the water level in the lake rise above its highest historical point. For example, in 1991, the water level in Xishan Station of Taihu Lake reached a record 4.79 m by July 15 and stayed at this level until July 16, 14 cm above the highest level recorded in 1954.

Due to high-speed economic development in recent years, urban flooding protection facilities and rural embankment also developed greatly. The new dikes and flood-discharging stations have allowed the water-discharge capacity to rise, and it now exceeds 7,000 m^3/sec. However, the capacity for in-discharging and out-discharging has not balanced; the capacity to discharge water from watercourses to rivers and sea is only half that from diking areas to watercourses.

12.5.3. Driving Forces for Water Environmental and Aquatic Ecosystem Changes

The Great Increase in Pollution Emissions, the Polluted Drainage Entering Lakes, Resulting in the In-Equilibrium of Input and Output. According to statistics, every year the total wastewater discharge in the Taihu Lake Basin increases by up to 3.19 billion t: COD_{Cr}, 282,404 t; TN, 79,552 t; TP, 5,660 t. Every year the main pollutants flowing into the Taihu Lake are: COD_{Cr}, 131,233 t;

TN, 30,635 t; TP, 1,751 t. These especially P and N, apparently exceed the assimilative capacity of the Taihu wetland ecosystem.

The Losses and Imbalances of Ecological Processes by Irrational Industrial Development

(1) The destruction of lake wetlands ecosystems Lake wetland ecosystems in the basin are the filter areas of lakes, which serve to deposit, absorb, transport, and transform the input and output of organic matter such as N and P, etc. Artificial dams, breakwaters, dykes and other industrial activities along the Lake wetlands lead to serious degradation and destruction of the Taihu Lake ecosystem.

In 1981, there were still 66 species of vascular hydrophytes, belonging to 29 families and 49 genera, in Taihu Lake. The area of waterweeds is 14,446 ha, which occupies 6% of Taihu Lake. The yearly productivity for vascular hydrophytes is about 450,000 t, among which reeds are over 70,000 t, water oats (*Zizania latifolia*), about 225,000 t, and submerged hydrophytes 147,000 t. Every year, there are about 400,000 t of these aquatic plants harvested as feeds for pond fish farming, farm composts and materials for papermaking and knitting. Every year, over 2,160 t nitrogen, more than 750 t phosphor and much CO_2 are absorbed, transformed and accumulated through these plants. However, many waterweeds in Taihu Lake, and other lakes in the basin, are now severely damaged and displaced: vascular hydrophytes in west Taihu Lake and other originally weedy lakes have vanished or are being eliminated. Thus, the links and routes of wetland ecosystem processes are interrupted and even broken for transporting, transforming, accumulating and balancing the output and input of aquatic inorganic nutrients and organic matter through vascular hydrophytes. Moreover, the competition between the main aquatic plants (as submerged hydrophytes) and aquatic algae is undermined. Therefore, more nutrients in lakes are utilized by algae. The biomass and production of algae increases abruptly, leading to considerable eutrophication.

(2) The impacts of industrial development on the lake ecosystems In the Taihu Lake Basin, the intensity of various developing activities, especially landuse, is heavy: it is estimated that, as a result of fish-farming alone, the TP and TN flowing into the lake are 102 and 527 t every year, respectively. Inappropriate fishing methods and lake surface shipping has also damaged the aquatic ecosystem in the lake.

12.5.4. Driving Forces of Engineering for Wetland Ecosystem Changes

(1) The conflicts between increasing amounts of pollution emission and the great lack of facilities for wastewater treatment have exacerbated the negative changes to wetland ecosystems in the Taihu Lake Basin. For instance, the treatment capacity of concentrated domestic wastewater treatment plants built in Suzhou, Wuxi, and Changzhou city totaled only 230,000 t/day, less than 1/4 of the municipal wastewater, discharge in Jiangsu province. The insufficient capacity of industrial wastewater treatment, the low efficiency of treatment operations, and the poor rate of up-to-standard treatment mean that pollutant discharge into the lakes is non-stop.

(2) Reclamation projects such as dyke-making and blocking up the water flow have changed landuse patterns that altered ecological hydrological processes. These have reduced the capacity to regulate hydrological processes (such as water level and water storage) in some sections of large or middle-size lake wetlands. For example, reclamation in 1976 blocked the five waterways for outflow connecting through Zhangwandang marsh on the lower reach of Taihu Lake. Because of this, the water from the upper reach of the lake via this marsh, originally discharging water to the north, now flowed south and elevated the water level by 0.2 m in the north Jiaxing areas in the upper reach of the lake.

According to the statistical data in 1990, there are about 7,000 dykes in the Taihu Lake, having a total area of 11,000 km^2, of which 498 dykes with a total area of 528.55 km^2 were developed within the period 1949–1985 (Table 13). The number of lakes for reclamation has reached 239 (33.8% of the total area of lakes) since 1949. The number of lake ecosystems that have disappeared due to reclamation is 165, which is 23.3% of the total lakes in the region (Table 14). Dyking in some lowlands made some lakes enclosed. They thus became inner ports and inner lakes so that certain ecosystems processes, such as nutrient recycling, were stopped. Although the water existed in dyked areas, the capacity to regulate and balance ecological processes, such as mitigating floods in the basin, had been lost (Nanjing Institute of Geography & Limnology, 1988; Wang, 1994; Han & Mao, 1995).

(3) As the impermeability of land increases with the increase in urban landuse, so the ground runoff coefficient becomes correspond large, which alters the ecological hydrological processes. For example, during the flooding period in 1991, the runoff coefficient of the Wujing-Yangcheng-Wuxi areas that have many towns was 0.758, but the runoff coefficient in the areas along Taihu Lake was 0.664. In addition, the area of some natural waterbodies decreased to a large extent because engineering for urban expansions caused some original rivers and lakes to be filled and leveled up. In Changzhou city, due to urban

Table 13: Statistical data on reclamation dynamics by different zones.

Name of lakes	1950s		1960s		1970s		1980s		Total	
	Number of dykes	**Area (km^2)**	**Number of dykes**	**Area (km^2)**	**Number of dykes**	**Area (km^2)**	**Number of dykes**	**Area (km^2)**	**Number of dykes**	**Area (km^2)**
Taihu Lake Zone	7	9.23	39	67.73	68	82.16	2	1.05	116	160.17
Yaoge Lake Zone	3	2.01	28	28.63	91	147.32	4	4.18	126	182.14
Hangzhou-Jiaxing-Huzhou Zone	2	0.37	14	10.76	54	24.94	6	2.76	76	38.83
Dianmao Zone	1	0.45	26	17.36	84	58.63	5	2.05	116	78.49
YangchengZone	2	0.94	8	28.08	24	19.05	2	0.30	36	48.37
Jiashan-Pinghu Zone	1	0.35	10	4.41	13	8.91	4	6.88	28	20.55
Total	16	13.35	125	156.97	334	341.01	23	17.22	498	528.55

Table 14: The dynamics of reclamation in large or middle-size lakes in Taihu Lake Basin.

Names of lakes	Lake size (km^2)	1950s			1960s			1980s			Total		
		Number of dykes	The area of built dykes (km^2)	Conserved lake area (km^2)	Number of dykes	The area of built dykes (km^2)	Conserved lake area (km^2)	The area of built dykes (km^2)	Conserved lake area (km^2)	The number of built dykes	Conserved lake area (km^2)	The number of built dykes	The area of built dykes (km^2)
Taihu Lake	2,587.98	7	9.23	2,573.75	39	67.73	2,511.02	68	2,428.86	2	2338	116	160.17
Gehu Lake	253.78	–	–	253.78	19	22.71	231.07	49	146.37	–	146.37	68	107.41
Yang cheng Lake	122.87	1	0.10	122.77	–	–	122.77	5	119.13	1	119.03	7	3.84
Yaohu Lake	111.43	–	–	111.43	1	0.65	110.78	19	90.47	2	88.97	22	22.46
Dingshanhu Lake	65.32	–	–	65.32	1	0.80	64.52	–	64.52	1	63.80	2	1.52
Chenghu Lake	44.42	–	–	44.42	–	–	44.42	3	40.64	–	40.64	3	3.78
Total	3,185.80	8	9.33	3,176.47	60	91.89	3,084.58	144	2,889.99	6	2,886.62	218	299.18

expansion, half the small ponds have been filled up, and the area of rivers in the urban districts decreased to only 7.36 km^2. This has weakened the capacity of urban wetland ecosystems to regulate and balance ecological hydrological processes, and reduced their ecological services, such as flood control or mitigation (Sun et al., 1993).

12.5.5. Driving Forces of Management for Wetland Ecosystems Changes

The imperfect management of wetland lake ecosystems, such as through unsound laws and regulations, worsens the water pollution and speeds up degradation of lake wetland ecosystems, especially eutrophication in Taihu Lake. According to the investigation, in some developed cities such as Suzhou, Wuxi and Changzhou, the annual economic loss caused by water pollution will occupy 5–7% of the GNP. In 1993, the immediate loss due to environmental pollution in Shanghai was around RMB 6.2 billion, accounting for about 3% of the GNP of the city, which greatly exceeded the investment on environmental protection. In the basin, the annual economic loss due to water pollution is at least RMB 5 billion. Therefore, weak ecosystem management has become a great constraint with regard to solving the problem of environmental degradation of wetland ecosystems in the Taihu Lake Basin.

12.6. Integrated Human Responses to Building Sustainable Security for Ecosystems of the Taihu Lake Basin

To realize sustainable development of the Taihu Lake Basin, it is necessary to establish the perfect human mechanism with which to manage the human–water interactions in the ecosystem of the Taihu Lake Basin, and to promote or enhance the services of wetland ecosystems as water resources, water quality, and water culture. We now discuss integrated management responses to changes in the ecosystems of the Taihu Lake Basin, using the HREEES conceptual framework.

12.6.1. Human Responses I (Water Resources): Sustainable Development and Uses of Water Resources

The overall strategic responses of sustainable development, uses, protection and recycling of water resources are: developing new forms of water resources, saving water, and regulating the storage capacity of reservoirs to increase availability; at the same time, implementing strict measures to protect and improve water quality, strictly control the overall volume of wastewater emission, and increase the reuse and recycling ratio of utilized water.

Developing Various Forms of Water Resources and Increasing the Availability of Water

(1) Developing various forms of water resources to withdraw water from rivers To better meet the increasing demand for water, it is necessary to develop new forms of water resource and increase the utilization ratio of water resources. The main measures are as follow: to establish water transport facilities; to promote and strengthen implementation of water conservancy projects along the rivers in south Jiangsu Province; and to develop the water withdrawal works of big rivers. These water withdrawal works mainly include two activities: (i) the renewal or renovation of old water withdrawal facilities from the Yangtze River and the dredging of the silted waterways to expand the total capacity of water withdrawal from Yangtze River and (ii) the implementing of the Fu-chun-jiang River Water Withdrawal Project in Zhejiang province.

The Taihu Lake plain has advantages in its natural geographic conditions, with the Yangtze River in the north and the Qiantangjiang River in the south. The water resource in the Yangtze River is rich. Normally, 1–1.5 billion m^3 of water can be withdrawn per year from the Yangtze River. If a new project at Yuhe is finished, then water will be able to be drawn from the Yangtze River at a rate of 150 m^3/sec in periods of high water demand. The imbalance between demand and supply can be mitigated by increasing the quantity of water drawn.

(2) Protecting and utilizing the high quality of water resources There exist some areas with high quality water resources in the Tiaoxi drainage area to the southwest of the Taihu Lake Basin. It is the best quality water within the Taihu Lake Basin, meeting Class II of the national standard GB 3828-88. The precipitation of the Tiaoxi drainage area is high, being 1,450 mm annually, which is about 1.5 times the average for the Taihu Lake Basin. If good quality water resources are well protected, as much as 0.4 billion m^3 of water could be supplied to meet the increasing demand in the adjacent cities.

(3) Sustainable exploitation, usage, and the control of underground water resources A Water Resource Administration Council should be established to enhance the sustainable management of underground water resources, to prevent excessive extraction of underground water resources, and to ensure the sustainable exploitation, usage and protection of underground water resources. Very efficient water supply systems should be established in newly developed areas of a city. In areas which have suffered from excessive extraction, proper measures should be taken to establish an effective supervision system for preventing subsidence of the land surface and the recharging of ground water into the underground water system (Yan & Zhang, 1997).

(4) Sufficiently protecting, reusing and recycling of limited water resource, and preventing water pollution Water pollution leads to low quality water, which greatly affects the availability of water resources. Since the ground water situation in cities is even worse, protecting water quality, preventing and controlling pollution, and fully reusing or recycling the limited available water are the main ways to improve the quality of local water, and to resolve the contradiction between supply and demand (Yan, 1994).

Increasing the Water Storage Capacity of Rivers, Lakes and Reservoirs, and Regulating the Annual and Seasonal Changes of Water Resources. The water storage of lakes in the Taihu Lake Basin reaches 6.8 billion m^3 with 72.8% in large and medium sized lakes. There are over 500 small dykes formed by blocking about 500 km^2 of the lake. This has directly reduced the area, and thus the capacity of lakes for water storage. Blocking lakes should be strictly limited, and some blocked fields of lakes should be returned to lakes. For deeper areas of lakes with sediment silt, the capacity to store water can be increased by dredging, which removes mud and eliminates obstacles from the river beds.

Promoting Measures for Saving Water in Agriculture, Industry and Cities. The international and domestic technology of water-saving in agriculture and industry should be promoted. Saving water in industry should be first realized by increasing the reuse rate of utilized water, as well as by establishing recycling systems for wastewater in order to improve the water environment. Water-saving engineering in agricultural irrigation can be developed by popularizing advanced irrigation technology, and by improving water use management.

12.6.2. Human Responses II (Environment): Protection of the Water Environment

Defining the Overall Amount of Control for Pollution Emissions. The annual emission in the Taihu Lake Basin has reached over 3.5 billion t per year, 80% of which is not treated: even if treated, it does not reach the State standard for emission to rivers and lakes. Used water flowing into and out of Taihu Lake should meet class III standard for ground water. It is essential for us to define the overall amount of control (Table 15) (Qin & Chen, 1996).

Maximum amounts of emission to be allowed in the Taihu Lake Basin in 2010 are as follows: TP 2,285 t/yr; TN 9,635 t/yr; COD_{Cr} 185,234 t/yr (Table 16). The maximum amounts of emission allowed by Zhejiang and Jiangsu Provinces are listed in Table 17.

Table 15: Targets for overall pollution amount control.

Controlled area	Index	Present emission (t/yr)	Maximum allowed emission (t/yr)	Maximum amount allowed to enter Taihu Lake (t/yr)	Province
Wuli lake-Meiliang lake seriously polluted control areas	COD_{Cr}	48,838	21,136	8,742	Jiangsu Province
	TP	1,225	319	91	
	TN	17,549	1162	412	
West Taihu Lake lake pollution controlled areas	COD_{Cr}	60,570	44,238	22,844	Jiangsu Province
	TP	1,478	859	245	
	TN	17,528	1,537	545	
West of Zhejiang Province pollution controlled areas	COD_{Cr}	56,784	38,614	16,023	Zhejiang Province
	TP	982	336	101	
	TN	14,120	1,131	401	
Wangyu River pollution controlled areas	COD_{Cr}	27,623	11,049	4,585	Jiangsu Province
	TP	671	204	58	
	TN	7,661	612	217	
Lakeside pollution controlled areas	COD_{Cr}	18,298	11,894	435	Jiangsu Province and Zhejiang Province
	TP	651	190	54	
	TN	11,926	1148	407	
East Taihu Lake–Taipu lake pollution controlled areas	COD_{Cr}	11,919	5791	2403	Jiangsu Province, Zhejiang Province and Shanghai city
	TP	161	133	38	
	TN	3,233	522	185	
Total	COD_{Cr}	22,4032	13,2722	59,532	
	TP	5,168	2,041	587	
	TN	72,017	6,112	2,167	

Note: The target of total amount control is allocated to the emission management authorities of subordinate district by every province in different periods.

Table 16: Maximum allowed amount of emission and amount into Taihu Lake in 2010.

Control index	Maximum amount into Taihu Lake	Maximum amount of emission
TP (t/yr)	719	2,285
TN (t/yr)	5,483	9,635
COD_{Cr} (t/yr)	90,761	185,234

Note: The maximum amount of emission allowed in 2010 includes precipitation, dust (sources unable to control): TP-93t/yr; TN-3181t/yr; CODCr-23,595 t/yr.

Zoning Polluted Water Areas by Protection Requirements. In order to control pollution and eutrophication of Taihu Lake, it is important to zone the polluted water areas by protection requirements and to determine the priority protection areas. The water areas in Taihu Lake can be zoned by protection requirement as follows (Qin & Chen, 1996):

- Meiliang Lake (including Wuli Lake): 135 km^2
- Gonghu Lake: 147 km^2
- East Taihu Lake: 158 km^2
- Water areas near Xukou area: 150 km^2

Considering the distribution of pollution loading and the regional targets for water quality protection, the following six areas can be identified:

- Meiliang Lake–Wuli Lake serious pollution control areas
- West Taihu Lake pollution control areas
- West Zhejiang province
- Wangyu River pollution control areas:
- Lakeside pollution control areas
- East Taihu Lake–Taipu River pollution control areas.

Table 17: Maximum amount of emission allowed by each province in 2010.

Control index	Jiangsu Province	Zhejiang Province	Total
TP (t/yr)	1,774	418	2,192
TN (t/yr)	5,163	1,291	6,454
COD_{Cr} (t/yr)	112,239	49,400	161,639

These six areas of water quality protection occupy more than 80% of Taihu Lake. The TP from the six areas of water quality protection reaches 5,168 t/yr (91% of the total amount in Taihu Lake); TN is 72,017 t/yr (91%); and COD_{Cr} is 224,032 t/yr (79%). The characteristics of each area of water quality protection are listed in Table 15.

As protection objectives for water quality in the year 2010, the following will be implemented: keeping the quality of water in main rivers flowing into and out of Taihu Lake up to class II of National Standard GB38338-88, and the maximum amount of pollution emission at TP 2,041 t/yr, TN 6,112 t/yr, COD_{Cr} 132,722 t/yr.

Controlling Pollution Sources and Reducing the Total Amount of Wastewater and Other Pollution Emissions

Targets of Pollution Source Control. The main targets of pollution source control are as follows:

(1) Wastewater discharged into the main streams, the first-class branches and the source areas for drinking water should meet the State's standard of Class I, and the wastewater that is discharged directly into Taihu Lake should meet the local standard if it is stricter than that mentioned above.
(2) Wastewater discharged into the areas of IV and V should meet the standard of Class II.
(3) Wastewater discharged into the urban sewage systems needing secondary treatment should meet the standard of Class III.

Ensuring That the Industrial Pollution Sources Meet Their Appropriate Standards. Some necessary measures include: (1) Adjusting the industrial structure to reduce the discharges of wastewater and other pollutants; preventing projects producing heavy pollution from being started; and where current enterprises produce heavy pollution that cannot be improved to the State's standard, then these enterprises should be closed, stopped, combined, transferred or relocated; (2) Introducing the new procedures and technologies to control the wastes generated; (3) Promoting reuse and recycling of the raw materials in the life cycle of the production; (4) Treating and recycling urban wastewater; (5) Promoting non-point pollution source control in farmland; some non-point pollution from fertilizer and pesticides should be strictly controlled; (6) Improving non-point pollution control of lakes by replacing integrated netted fishery in drinking water withdrawal sites such as Wuli Lake, Meiliang Lake, Gong Lake, East Taihu Lake (and Xukou nearby areas); and also by stopping mechanized fishing, changing tourism boats into electricity-driven ones and so on.

12.6.3. Human Responses III (Wetland Ecosystems): Improving Structure and Function of Wetland Ecosystems, Rehabilitating Disturbed or Destroyed Ecosystems, and Improving Their Ecological Capacity of Services

Based on some ecological principles such as dissipation, hierarchy, recycling and self-organization, improvement in the structure and function of wetland ecosystems can be implemented in various ways such as: increasing loops of material recycling in chains and webs; linking parallel loops or processes into self-organized systems; using or reusing products, by-products and wastes through a hierarchy of layers and levels of efficiency; and restoring the structure and functions of destroyed or degraded wetlands ecosystems, etc. The purposes of these efforts are to improve ecological services of self-cleaning, self-regulating, self-maintenance and self-organization, and to enhance multi-production of different-level products simultaneously (Hu, 1997).

12.6.4. Human Responses IV (Wetland Ecosystems Engineering)

One of the most important human responses is the ecosystem engineering approach, in order to improve or enhance the dynamics of wetland ecosystems. Some key ecosystem engineering pilot projects should be set up to realize the purpose of ecological restoration or enhancement in the Taihu Lake Basin. Eight pilot projects such as agricultural non-point pollution source control, ecological agriculture, sedimentary mud dredging projects, etc., are included (Table 18) with a total investment of RMB 54 million.

12.6.5. Human Responses V (Ecosystems Management): Building a Modern Ecological Culture for Realizing the Sustainable Management of Wetland Ecological Security

One of the key actions for promoting the transition of management towards sustainable secure water resources, water environment and wetland ecosystems is the modern ecological culture developing in the Taihu Lake Basin (Hu & Wang, 1997). Some of the main points of this are as follow:

Promoting Institutional Transition of Wetland Ecosystem Management for Improving Efficiency, Equity, and Wetland Ecosystem Security Management in the Taihu Lake Basin

(1) Provincial or city governments in the basin should be responsible for wetland ecosystem security and develop strict regulations, backed by legislation, as a

Table 18: Key ecosystem engineering pilot projects in the Taihu Lake Basin.

Items	Investments RMB 10^4	Remarks
Farming waste recycling project	400	First protection area of the west parts along with Taihu Lake
Proposd reservior project	450	First protection area of the west parts along with Taihu Lake
Big aquatic vegetation restoration project	700	Shallow water area of the northeast parts along with Taihu Lake
Algae collection and integrated utilization project	300	Meiliang lake area
Ecological agriculture project	500	Zhejiang province
Bank protection belt project	750	First protection area of the west parts along with Taihu Lake
Sedimentary mud dredging project	300	Dredging up 100,000 m^3 of bottom mud
Cleaning production project	2,000	Zhejiang and Jiangsu provinces

priority for action to improve the management and utilization of wetland ecosystems.

(2) Promoting price system reform for water resource uses in the basin. Currently the water price is too low to make local people change their old behavior of using water indiscriminately. Adjusting the water price can strengthen the public consciousness for saving water.

(3) Governments at different levels should further promote an ecological culture of recycling, reuse, retrieval and reduction, by implementing industrial structure adjustments and by conducting demonstrations of clean production projects.

(4) All of the developmental projects must be matched with environmental impact assessment.

(5) Stringent control should be in place to reduce agricultural contamination. Efforts should be made to popularize ecological agriculture. Superfluous use of fertilizer should be limited in an economic way. The use of organic fertilizer should be encouraged. Chemical products containing nitrogen and phosphorus which contribute to eutrophication of lakes should only be used in limited amounts in the basin.

(6) Promoting the transition of the traditional model of citizens' consumption into sustainable consumption. Domestic discharge of nitrogen and phosphorus from wastewater should be reduced in an effective way. Use of detergents

containing phosphorus should be banned in first- and second-class protection areas in the Taihu Lake Basin. In the third-class protection areas it could be retained in a limited amount.

Redirecting Economic Capital to Invest in Ecologically Healthy Products, and the Enhancement of Ecosystem Services. Economic investments need to be redirected from the single objective of profit maximization to a joint objective of maximizing both economic profits and ecological services. Environmental or ecological projects in the Taihu Lake Basin should be carried out jointly with infrastructure construction and other industrial development. According to the current estimate, the total investment on priority projects for water environmental protection and ecological building is RMB 20.4 billion (Table 19) (Qin & Chen, 1996).

Table 19: Investment of priority projects for water environment and ecosystems in Taihu Lake (RMB 100 million).

Engineering types	**Jiangsu Province**	**Zhejiang Province**	**Shanghai city**	**Total**	**Rate (%)**
Wastewater treatment projects (including hotels and restaurants)	52.50	27.84	0.2	80.54	39.45
Industrial source abatement projects	2.88	2.59	0.021	5.50	2.69
Non-point source pollution treatment projects	5.44	3.16	–	8.60	4.21
Inner source pollution treatment projects	7.04	–	–	7.04	3.45
Water conservancy and drinking water guarantee projects	15.2	14.19	7.38	36.77	18.0
Pollution interception and river channel treatment projects	12.13	24.06	–	36.19	17.72
Clean production projects	16.75	12.25	–	29.0	14.20
Pollution treatment pilot projects	0.34	0.20	–	0.54	0.26
Total	112.29	84.30	7.60	204.18	100
Rate (%)	54.99	41.28	3.72	100	

Promoting Research on Key Engineering and Management of Wetland Ecosystems in Taihu Lake Basin. To realize wetland ecosystem security, it is necessary to promote scientific research, especially on eutrophication of lakes, and to develop key ecological technologies for solving the degradation of lake ecosystems. The topics, which need be tackled as key scientific projects include:

(1) Agricultural non-point source pollution technology in river networks areas
(2) Aquatic vegetation restoration
(3) Sedimentary mud dredging and secondary pollution prevention
(4) Monitoring network technologies, including warning systems for protection of important and sensitive areas, and decision-making support systems for integrated management of the Taihu Lake Basin
(5) Gross-amount pollution control technologies
(6) Life-cycle assessment for products, enterprises and industrial parks
(7) Ecological mechanism of "water bloom" in Taihu Lake
(8) Sewage interception technology in Wuli Lake and Meiliang Lake
(9) Pollution control for animal breeding

Promoting Ecological Education to Enhance Ecological Thinking in the General Population. The transformation of awareness and behavior from interest-and-reputation dominated to ecological-norms dominated is a key factor in ecological education to promote sustainable ecosystems management (Hu & Wang, 1997). Thus, encouraging the population to think in terms of ecological norms, morality, values and beliefs needs to become an important part of ecological education. Information technology, other high technology, and ecological industrial technology need to be integrated in order to contribute to ecological education: that is to say, to promote ecological educational through technology integration and to develop modern technology through ecological education. This will be beneficial for rebuilding technology, engineering and institutions, and may play a positive role in finally establishing sustainable ecosystem management in the basin.

12.7. Conclusions

Using the concept of HREEES Systems, this research has made an integrated analysis of human–water interactions in lake wetland ecosystems of the Taihu Lake Basin, East China. The DPR (driving forces–states–responses) methodological model is used to analyze the human driving forces of urbanization and industrialization, which cause changes in the resources, environment, ecosystems, and security of lake wetland ecosystems in the Taihu Lake Basin. The analyses has shown that landuse, water resource exploitation and environmental impacts or disturbances from engineering and management activities are key driving forces for the changes of wetland ecosystems in the structure and functions in the Taihu Lake Basin. Thus, the two critical transitions of mechanisms, from ordinary environmental engineering into ecological engineering and from typical end-of-pipe management into life-cycle-oriented ecological management, are leading factors required to enhance the security of the Taihu Lake wetland ecosystems. Integrated responses to wetland ecosystem changes in the Taihu Lake Basin should be focused on building an ecological basis for engineering and management. Scientific research and the development of a modern ecological culture of management is necessary to direct human–water interactions towards sustainable co-evolution in the lake wetland ecosystems of the Taihu Basin.

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Chapter 13

Ecological Benefits of Italian Poplar Afforestation in Wetland Areas along the Yangtze River, Fanchang County of Anhui Province

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Abstract. In the wetlands along the Yangtze River there are some bank areas where the water usually accumulates, or overflows, about 90–120 days during the rainy season from May to August, and low areas are covered with water all the year. Fanchang County is located on the mid-lower reaches of the Yangtze River in Anhui Province. Within its boundary, there are about 1000 ha of wetlands along the river, associated with eight towns. In 1990, three new varieties of Italian Poplar Tree (*Populus deltoides* cv. (Harvard), *P. deltoides* cv. (Lux), and *P.* × *euramericana* cv. (San Martian)) were introduced to the county, and 64 ha were afforested. By the year 2002, 540 ha had been planted across the county and 28 ha harvested.

This chapter deals with some social, economic and ecological benefits from poplar tree afforestation at wetlands along the Yangtze River in Fanchang County of Anhui Province. Some ecological problems from poplar tree afforestation are also analyzed, and future strategies suggested.

13.1. Introduction

Wetlands have important ecological functions, not associated with other types of ecosystems (Mitsch & Gosselink, 2000; Keddy, 2000). The wetlands along the Yangtze River include the floodplain areas regularly influenced by flood water, and some bank areas along the river. In the rainy season from May to August, the water-table level rises, so that the floodplain and some bank areas are submerged for about 90–120 days. The wetlands along the Yangtze River include three

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components: the embankment swamp, the floodplain areas, and the other mixed types of wetlands. The embankment swamp usually refers to an area always existing between the floodplain and the bank.

The wetlands along the Yangtze River in Anhui Province are associated with several cities including Anqing, Chizhou, Tonglin, Fanchang and Maanshan. The total area of wetlands is about 13,330 ha, of which 8000 ha are floodplain, and 5,330 ha are other wetlands inside the bank (Department of Forestry, Anhui, 2000). Fanchang County is located on the mid-lower reaches of the Yangtze River in Anhui Province. Within its boundary, there are about 1000 ha of wetlands along the river, associated with eight towns. In 1990, three varieties of fast-growing, water-tolerant Italian Poplar Tree (*Populus deltoides*cv. (Harvard), *P. deltoides* cv. (Lux), and *P.* × *euramericana* cv. (San Martian)) were introduced into two towns of the county, and the area of afforestation was 64 ha, which was 0.48% of the total wetland area in Anhui Province. The three fine varieties of artificial breeding are seedling progeny from the free pollination of black American poplar and Euro-American poplar crossbreeded, respectively. They all possess the following characteristics: (1) They are adapted to grow in the warm and humid environment. The average temperature is about 14°C, with sunshine duration more than 1400 h per year and an annual rainfall ranges from about 1200–1400 mm, especially areas along the banks of the Yangtze River; (2) They are cultivated species, with three varieties: photophilic, hygrophilous and oxyphilous; (3) They are not resistant to cold nor wind; and (4) They are readily subjected to attack by pests and diseases. Up to the year 2002, 540 ha had been afforested with hybrid poplar across the county, which is 4.05% of the total wetlands area in Anhui Province (Fig. 1), and by 2001 only 28 ha hybrid poplar had been harvested.

This paper analyzes the social, economic and ecological benefits of afforestation with poplar trees of wetland areas along the Yangtze River in Fanchang County, Anhui Province. In addition, the main ecological problems arising from afforestation with poplar trees are discussed, and finally some sustainable afforestation strategies are suggested.

13.2. The Wetlands along the Yangtze River, Fanchang County

13.2.1. General descriptions

Fanchang County is located on the south bank of the Yangtze River in Anhui Province, 118–118°20′E and 31–31°18′N. It belongs to the northern subtropical and eastern humid monsoon circulation zone. There are about 1000 ha of wetlands along the river, which is 7.5% of that in the province. The wetlands form a strip along the bank, associated with the following towns: Digang, Lunan, Xingang,

Figure 1: The landscape of 36 ha of Italy Poplar trees "*Populus deltoides* cv. (Harvard), *P. deltoides* cv. (Lux), and *P.* × *euramericana* cv. (San Martian)" in the floodplain wetland, Xiaozhou township, Fanchang County, Anhui Province.

Gaoan, Sanshan, Xiaozhou, Baoding and Zhonggou. There are 3400 households and 16,500 residents in the areas along the bank. There are also stands of willows on the floodplain areas outside the bank (Fig. 2). The soil is podzolic, some of which is silty. In its section plane, the topsoil is sandy, about 30–120 cm in depth; the middle layer is discontinuous podzolic silt loam; and the lowest layer is also podzolic silt loam. The composition of the soil is: organic matter, 1.01%; total nitrogen, 0.077%; available phosphorus, 5 mg/kg; available potassium, 60 mg/kg. The nutrients in the soil mainly come from soil eroded by flood water in the upper reaches of the Yangtze River. During the low water season, the plant species in

Figure 2: The wetlands of willow trees along the bank of Yangtze River.

floodplain areas are: *Eleusine indica*, *Alternanthera philoxeroides*, *Polygonum* spp., *Imperate cylindrica* var. *marjor*, *Cyperus* spp., *Caris* spp., *Salix matsudana* Koidz., etc. Species in wetlands within the bank are: *Phragmites communr*, *Micanthus sacchariflоorus,* etc. In the swamps along the bank, the plants are: *Eleocharis* spp., *Phragmites* spp *Micanthus saccharifloorus*, *Equisetum* spp., etc., and bird species are: *E. garzetta*, *Amaurornis phoenicuru*, *Sterna hirundo*, etc. (Anqing City, 2002).

13.3. Characteristics of Wetlands in the Floodplain Areas

The floodplain is located between the river and hilly land, with a high groundwater level, and is covered by flood water during the growing season. The interactions among the river, the floodplain and other hilly land have formed the following five characteristics:

(1) The floodplain stretches like a belt along the river.
(2) The floodplain wetland is an open ecosystem with different ecological functions; it is neither hilly land nor aquatic ecosystem.
(3) The catchment area in the upper reaches of the river is an important factor influencing the flow volume, flow velocity, and the duration time of floods in the lower reaches. The larger the area of catchment, the greater is the flow volume; and the wider the river, the longer is the duration of the flood. If the flow velocity changes are small, the possibility of flooding is less (An, 2003).
(4) The floodplain is submerged in water for about 90–120 days during the rainy season, from late May to the middle of September.
(5) The species diversity of plants and animals change seasonally.

13.4. Ecological Benefits of Poplar Trees Afforestation

13.4.1. Turning Reed Wetland into Poplar Forests Reduces the Density of Oncomelania hupensis in Floodplain and the Incidence of Schistosomiasis in the Floodplain

In 1990, the government in Anhui Province had implemented a project for "developing forestry and controlling *Oncomelania hupensis* infection". The purpose was not only to enhance wetland ecosystems along the river, and to turn the floodplain into healthy wetland, but also to reduce the density of *O. hupensis* in afforested areas. According to relevant data for the province (Anqing City, 2002; The State Redactal Team of Total Report, 1994), it is

Table 1: The distribution of *Oncomelania hupensis* in Italian Poplar stands in Digang, Xiaozhou, Fanchang County and Xinzhou, Danyang City.

County/ town	Number of investigated cases	Number of living *Oncomelania hupensis* cases	L.R. (%)	Living *Oncomelania hupensis* (averaged density ($O./0.11\ m^2$)	Infected *Oncomelania hupensis* averaged density ($O./0.11\ m^2$)	Afforested period (years)
Digang	281	13	4.6	1.4	0.0006	4
Xiaozhou	290	13	4.4	1.2	0	4
Xinzhou	375	12	3.2	0.7	0	2

L.R. indicates the ratio of the cases with living *Oncomelania hupensis* to total cases; Zhang et al. (1992)

found that the appearance rate of living *O. hupensis*has decreased by about 80%, and the density of living *O. hupensis* by about 88.8%, with the infection rate nearly zero.

In the floodplain along the bank in Fanchang County, the hybrids black American poplar (*Populus deltoides* cv. (Harvard), *P. deltoides*cv. (Lux)), and Canadian poplar (*P.* × *euramericana* cv. (San Martian)) were first introduced to the townships of Digang and Xiaozhou. The floodplain in Digang township and the reed wetlands in Xiaozhou township have greatly improved with regard to afforestation and control of Schistosomiasis. During the flood season, the stand of poplar trees can control the dispersion of *O. hupensis*, and there are no infected *O. hupensis* (Tables 1 and 2).

13.4.2. The Ecological Roles of Italian Poplar in Mitigating Flood

In the rainy season, from May to August, the water table rises in the Yangtze River, and the speed and volume of the water flow is greater, hence its erosive power is also greater. The waves go beyond the floodplain and erode the bank directly, which is a great threat to the safety of the bank. In the past, the peasants who lived along the bank planted willow trees spontaneously every year. One of the purposes for doing so was to provide fuel; another was to reduce the eroding power of the water. So planting Italian Poplar trees in floodplains now reduces wind from the river, and decreases the erosive power of the waves (Table 3 and Fig. 3).

Table 2: The situation for detaining *Oncomelania hupensis* in Italy Poplar stands in Digang, Xiaozhou, Fanchang County.

Dates of investigation	**Places of investigation**	**Investigated plants/cases**	**Plants with *Oncomelania hupensis* cases**	**Total no. of captured living *Oncomelania hupensis***	**Percentage of plants with *Oncomelania hupensis***	**Average density of *Oncomelania hupensis* (*Oncomelania hupensis* per plant)**	**Percentage of infected *Oncomelania hupensis***
	Digang	100	23	41	41	0.41	0
1994.7	Xiaozhou	100	21	35	35	0.35	0
	Control plots	100	56	670	56	6.70	0
	Digang	100	17	21	21	0.21	0
1994.8	Xiaozhou	100	15	16	16	0.16	0
	Control plots	100	43	215	43	2.15	0
	Digang	100	2	2	2	0.02	0
1994.9	Xiaozhou	100	0	0	0	0	0
	Control plots	100	25	38	25	0.38	0

Table 3: Italian Poplar Afforestation in Digang and Xiaozhou, Fanchang County, Anhui Province.

Years	Tree height (m)	Breast height diameter (cm)	Crown extent (m × m)	Branch height (m)	Density (plants/ha)
			Italian Poplar trees in Digang		
1990	3	(3.5)*	–	–	840
1991	4	5.0	1.2 × 1.8	1	830
1996	18	15	4 × 5	3	830
2000	21	19	4 × 5	5	830
			Italian Poplar trees in Xiaozhou		
1990	3.5	(3.5)*	–	–	903
1991	4.2	4.2	1.5 × 1.7	1.7	900
1996	19.5	15.0	4.5 × 5	3.5	880
2000	22.0	20.0	5 × 5	6	800

* indicates the diameter of poplar seedling root.

13.4.3. Protecting the Biodiversity in Wetland Ecosystems along the River

The composition of species in wetland ecosystems was relatively simple. After the afforestation with Italian Poplar trees, the number of species has increased substantially as indicated below.

(1) Main species in the period of high water level: *Oncomelania hupensis*, *Eulota* spp., *Lymnaea*, *E. garzetta*, *Anoplophora chinensis*, *Apriona germari*, *Batocera horsfiedi*;

Figure 3: The edge areas of Italian Poplar forest wetlands eroded by flood water in Summer, 2002.

Figure 4: Winter migrant birds, *Sterna hirundo* rising from the willow forests in the bank of the Yangtze River, Xiaozhou township, Fanchang County.

(2) Main species in the period of low water level:
Birds: *E. garzetta*, *Sterna hirundo*, *Crypsirina* spp., *Passer montanus saturatus*, etc.
Insects: *Anoplophora chinensis*, *Apriona germari*, *Batocera horsfiedi*, *Holcocerus vicarius* (Walker), *Zeuzera coffeae Nietner*, etc. (Hua et al., 1990).
Spiral mollusks: *Oncomelania hupensis*, *Eulota* spp., *Lymnaea*, etc.
Plants: *Polygonum* spp., *Imperate cylindrica* var. *marjor*, *Cyperus* spp., *Caris* spp., *Salix matsudana Koidz.*, *Micanthus saccharifloorus*, *Rumex acetosa*, *Monochoria vaginalis*, etc.

(3) By afforestation with Italian Poplar in the floodplain wetland, the structure and ecological functions of the wetland have been improved, and the diversity of the species has increased. Meanwhile, these changes had positive effects on enhancing ecological services of wetlands to local social and economic development (Department of Forestry, Anhui 2000) (Fig. 4).

13.5. Social and Economic Benefits

13.5.1. Intercropping to Enhance the Efficiency of Agricultural Landuse

In 1990, there were 64 ha of floodplain afforested in the whole county, of which 62% were in the highlands. During the low water season, the local peasants practised intercropping (wheat, coles, etc.) between the trees. This type of farming not only increases the fertility of arable land, but also raises the income of the peasants.

Figure 5: The raw materials harvested for industrial production from Poplar trees forest with main species of *Phragmites communis*, *Miscanthus sacchariflorus*, *Imperata cylindrica* var. *major*, etc.

13.5.2. Providing Raw Materials for Industrial Production

The Italian Poplar trees have many uses due to the fine quality of their timber. Besides being used as raw material for building and furniture, they can also be used as raw materials for light industry, such as papermaking, fiberboard, and matches. The Poplar trees in Digang were all logged in 2001, producing about 8.5 t/ha. Local peasants can earn RMB 13,500/ha (RMB 100 equals US$ 12), the equivalent of RMB 1,227/ha/year. In addition, until the fourth year, local peasants can obtain 1.5 t of fuel from Poplar forests, which can partly solve the local shortage of energy (Fig. 5).

13.6. Some Problems and Suggestions for Sustainable Afforestation

Some ecological problems have also appeared during recent years, so effective ecological engineering should be implemented in connection with future afforestation as follows:

(1) Podzolic soil is not suitable for retaining nutrients for crop growth, which reduces the growth rate of the trees. According to Table 3, there are two periods of growth at two planting sites: the first period was from 1991 to 1996, and the second was from 1997 to 2000. The data clearly shows that the growth rate in both areas in the second period was much lower than that in the first period. Therefore, it is necessary for us to improve the management of Poplar tree plantation during the second period, especially to enhance the soil fertility.

(2) *Protecting the forests from diseases and pests*. Diseases and pests, especially wood-eating pests, easily attack Poplar trees leading to the destruction of whole forests. Therefore, it is suggested that planters should daub the stems with pesticides two or three times every April and May during the first and second years of afforestation. Thus, the number of pests can be reduced, the quality of the timber improved, and more profit achieved. Meanwhile, protecting birds and reducing the number of leaf-eating pests should also be considered.

(3) For improving plantation engineering, the density of afforestation must take into account the differences in soil qualities and water levels. The density of the trees will influence the density of *O. hupensis*, the water flow volume and the flow velocity during the flood seasons. Thus, an overall adjustment of plantation engineering can be achieved by adjusting the density of afforestation according to the differences of soil and water levels, and the density of afforestation can be changed to 5 m × 5 m per plant, an equivalent of 400 plants/ha. At the same time, intercropping should be implemented during the afforestation in order to increase the productivity of the land.

13.7. Conclusion

Afforestation with Italian Poplar trees on the floodplain wetlands along the Yangtze River has produced great ecological benefits in recent years, which include: (1) Reduction of the density of *O. hupensis* in the shallow areas, and the incidence rate of Schistosomiasis; (2) Mitigation of floodwater from the Yangtze River in the rainy season. (3) Italian Poplar forest wetlands along the Yangtze River have increased biodiversity in the floodplain areas. Economic benefits have been achieved, so that Digang township has now harvested 28 ha of poplar timber, with profits of RMB 13,500/ha in 2001. This ecological engineering of wetlands in Fanchang County of Anhui Province has contributed to the transformation of local wetland ecosystems, so that they can now be managed for sustainable development and local socio-economic progress.

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Developments in Ecosystems, volume 1
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Chapter 14

Wetland Conservation and Management in the Philippines: Where are We Now? The Case of Seagrass and Mangrove

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Abstract. A significant portion of the Philippine coastal habitats is at high risk of being lost in the next decade. Seagrass and mangrove ecosystems in the Philippines, known to be very productive and exhibiting high biodiversity, are under severe stress from the combined impacts of human overexploitation, habitat destruction, pollution, sedimentation and general neglect There is an urgent need to conserve and manage these habitats in the country in particular and in Southeast Asia in general. The reason for management of the coastal resources is that they are a huge natural and economic resource in the country in terms of food supply, livelihood, other revenue and quality of the environment.

To effectively conserve and manage Philippine seagrass and mangrove, we should: (1) focus research on priority management issues (link science to management); (2) develop an integrated framework for action; (3) undertake an economic valuation of the resources and of relevant policy changes; (4) forge public–private partnerships to manage, use, and conserve seagrass and mangrove; (5) ensure a functional coordination among concerned agencies; (6) increase the content of seagrass–mangrove repositories; and (7) for mangroves, adopt some "wise" management options. To protect the larger coastal and marine environment, governments should: (1) localize sustainable development through sound governance; (2) ensure high quality scientific publication (shift from description to synthesis); (3) position S and T centrally in economic development policy; and (4) adopt the Integrated Coastal Area Management philosophy.

14.1. Introduction

A significant portion of the Philippine coastal habitats is at high risk of being lost in the next decade. This is also true for the ASEAN region (Association of

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Southeast Asian Nations comprising Brunei Darussalam, Indonesia, Laos, Malaysia, Myanmar, Philippines, Singapore, Thailand, and Vietnam) where about half of its coastal resources have either been lost or are severely degraded during the past 56 years (Chou, 1994; Fortes & McManus, 1995; Fortes, 2001) and the rate of degradation is increasing. Human impacts are the primary cause for most of these losses and these are increasing as human populations increase. There is rapid economic and human population growth-over 80 million in the Philippines, with the population, just like in Southeast Asia, doubling in the next 25–35 years (World Resources Institute, 1990). Infrastructure development along coasts is doubling at almost decadal rates. People extract about 60% of the country's animal protein from the sea. These changes will result in greater demands for coastal zone resources, especially quality seafood products and space (Wilkinson, 2002). Our experiences in the past show that an explosive population growth coupled with rapidly dwindling resources will bring about short-term economic development mostly at the expense of the environment. The overall result of this potential loss of coastal resources are issues and concerns, which will farther aggravate the social and economic conditions of the greater portion of the region's population making ecological concerns serious socioeconomic issues. Today these issues bring about problems with far-reaching effects that go beyond sociopolitical boundaries.

There is an urgent need to conserve and manage seagrass and mangrove habitats in the Philippines in particular and in Southeast Asia in general. The reason we must manage our coastal resources is that they are a huge natural and economic resource in the country in terms of food supply, livelihood, other revenue and quality of the environment. Management, which implies wise use and maintenance of the resource, is crucial to ensure the continuous productive stream of net benefits without inputs from humans (White and Cruz-Trinidad, 2001). The problem in the Philippines simply is that we are damaging and overexploiting all the coastal ecosystems, compromising their natural productivity to the point of doing permanent damage to the entire system.

14.2. Status of Philippine Seagrass and Mangrove Habitats

Seagrass beds are estuarine or sea floor areas dominated by a discrete community of flowering plants. These plants are with roots and rhizomes (underground stems), thriving in slightly reducing sediments and normally exhibiting maximum biomass when completely submerged. With about 18 species in the Philippines, they grow best in estuaries and lagoons where they are often associated, physically and ecologically, with mangrove forests and coral reefs, often forming the ecotone between these two divergent ecosystems. Seagrass bed, as an ecotone, mediates the structural and dynamic components of the neighboring ecosystems via control of material, water, and energy flows between them. More importantly, seagrass systems support a rich diversity of species from adjacent systems and provide primary refugia for both economically and ecologically important organisms. As such, seagrass habitats are sensitive to fluctuations because species coming from their neighboring systems encounter "marginal conditions" and are at the extremes of their tolerance levels to environmental alterations. This sensitivity makes seagrasses useful indicators of changes not easily observable in either coral reef or mangrove forest.

A global picture of seagrass distribution has long been known (den Hartog, 1970). Related studies (e.g. Fortes, 1988; Mukai, 1990) augment our knowledge on seagrass biogeographical affinities. However, there are still wide areas where the existence of seagrasses likely remains unknown (Green & Short, 2003). A priority identified by the participants at the Global Seagrass Workshop (St. Petersburg, Florida, 9 November, 2001) was to come up with a map, which would provide the actual location and coverage of the world's seagrass habitats, incorporating their status in the face of environmental change. It is a partial result of an ongoing initiative of the United Nations Environment Programme-World Conservation Monitoring Center (UNEP-WCMC) to develop a comprehensive global GIS dataset coming from distribution maps from multiple sources.

Green and Short (2003) gives the most updated compilation of the seagrass flora of 115 countries and the current geographical distribution of the plants. The first of its kind for seagrasses, this points to three important findings: (1) the centers of diversity both at national and regional levels, with a clear focus in Southeast Asia reaching up to southern Japan, and a second focus of diversity in the Red Sea and East Africa; (2) some species have clearly restricted ranges; and (3) some species are endemic to single countries. Australia (31 species) and the Philippines (18 species) have the highest levels of diversity. The pattern reflects a high similarity with the global distribution of corals and mangroves. Interestingly, for seagrasses, this pattern extends farther north and into the temperate waters of Japan and show a much wider global distribution into cold temperate waters. It has been suggested that Southeast Asia may have been a center of species accumulation (the "vortex model of coral reef biogeography"), a region where, due to favorable climatic conditions in the recent ice-ages, species have converged ("vicariance hypothesis," McCoy & Heck, 1976), or a center for species evolution with the combination of benign conditions and changing sea levels ("eustatic diversity pump model").

Seagrasses play a significant role in global carbon and nutrient cycling. Although there is considerable variance on the productivity values reported for seagrasses worldwide, the values range from 500–4,000 g C $m^{-2}yr^{-2}$. This range ranks the plants among the world's highly productive ecosystems. The quantity of seagrass carbon available for storage in the sediments represent some 0.08 Pg C yr^{-1} (1 Pg = 10^{15} g) in the ocean as a whole (or 12% of the total carbon storage in the ocean despite its 1% contribution to the total oceanic production (Duarte & Cebrian, 1996). Although seagrasses only occupy a small fraction of the world's nearshore waters, and the total area of seagrasses is likely to be less than 10% of the shallow water area of the world's continental shelves (continental waters to a depth of 200 m, about 25 million km^2), they potentially could cover from 500,000–1,000,000 km^2.

Mangroves, on the other hand, is a type of forest growing along tidal mudflats and along shallow water coastal areas extending inland along rivers, streams and their tributaries where the water is generally brackish. Mangrove trees dominate the mangrove ecosystem as the primary producer interacting with associated aquatic fauna, social and physical factors of the coastal environment.

The Philippine mangrove flora consists of 47 true mangroves' and associated species belonging to 26 families (Melana & Gonzales, 1996). True mangroves grow in the mangrove environment; associated species may grow on other habitat types such s the beach forest and lowland areas. The mangrove fauna is made up of shore birds, some species of mammals, (monkeys, rats, etc), reptiles, mollusks, crustaceans, polychaetes, fishes and insects.

In the Philippines seagrass and mangrove ecosystems are two of the most productive and biologically diverse in the world. Table 1 shows the primary productivity of some of the major marine communities of the world. These ecosystems rank second and third in productivity, the basis of their economic and environmental roles in the region (Fortes, 1995a, 2001). With coral reefs, they are the major support ecosystems in Southeast Asia and the rest of the tropical world.

Table 1: Primary productivity of some major marine communities.

Community type	**Primary productivity (grams carbon/m^2/yr)**
Mangroves	430–5,000
Seagrass beds (only)	500–4,000
Algal and seagrass beds	900–4,650
Coral reefs	1,800–4,200
Estuaries	200–4,000
Upwelling zones	400–3,650
Continental shelf waters	100–600
Open ocean	2–400

Table 2: Primary producers of the oceans: estimates of area covered, total net primary production (NPP) and amount of this production consumed by herbivores, decomposed and stored.

Primary producer	**Area (10^6 km^2)**	**Total NPP (Pg C yr^{-1})**	**Herbivory (Pg C yr^{-1})**	**Decomposition (Pg C yr^{-1})**	**Storage (Pg C yr^{-1})**
Oceanic p-plankton	332	43	24.4	14.7	0.17
Coastal p-plankton	27	4.5	1.8	1.8	0.18
Microphytobenthos	6.8	0.34	0.15	0.09	0.02
Coral reef algae	0.6	0.6	0.18	0.45	0
Macroalgae	6.8	2.55	0.86	0.95	0.01
Seagrasses	0.6	0.49	0.09	0.25	0.08
Marsh plants	0.4	0.44	0.14	0.23	0.07
Mangroves	1.1	1.1	0.10	0.44	0.11
Total		5.3	27.8	19.00	0.65

After Duarte and Cebrian (1996).

Compared to the other primary producers of the sea, seagrasses and mangroves contribute substantially to the total oceanic production (Table 2). Note that despite the small contribution of seagrasses (1%) to the total oceanic production, it contributes 12% of the total carbon storage in the ocean (Duarte & Cebrian, 1996). The stored fraction accumulates in the system, as they are not decomposed within a year. On the other hand, mangroves account for 17% of the total marine CO_2 uptake. Worldwide seagrasses occupy an area of about 600,000 km^2 while mangroves occupy about 1,100,000 km^2 (Duarte & Cebrian, 1996).

14.3. Seagrass and Mangrove Biodiversity

A vast array of plants and animals live in seagrass beds and mangroves of the Philippines. This is due to the rich nutrient pool and diversity of physical structures protecting young marine life from predators. Fish and shrimp are probably the most important components of the habitats, although coastal villages in the country derive their sustenance from other components of the habitats.

Table 3 gives a comparison of species diversity among the major coastal ecosystems in the Philippines (modified from DENR/UNDP, 1997). Next to coral reefs, seagrass beds have the highest biodiversity. As in the whole of Southeast Asia, the figures may be grossly underestimated due to the paucity of documented information. There are indications largely through observations and ocular

Table 3: Comparison of species diversity among the major coastal ecosystems in the Philippines.

Taxon	Seagrass beds	Coral reefs	Soft bottoms	Mangroves
Seagrass	18	14	3	5
Algae	154	1,043	0	72
Corals	8	381	0	0
Other inverts	73	1,485	67	39
Fish	218	1,030	2	241
Mammals	1			
Reptiles	11	14		16
Total	483	3,967	72	373

Modified from DENR/UNDP (1997).

surveys, however, that the species richness in the habitats, particularly the fish and invertebrates, could be much higher than previously thought.

14.4. Threats to Seagrass and Mangrove

Seagrass and mangrove ecosystems in the Philippines are under severe stress from the combined impacts of human overexploitation, habitat destruction, pollution, sedimentation and general neglect (Chou, 1994; Fortes, 1995b). The resulting losses, expressed in thousands of dollars per year per km^2 of coastal area lost, have their greatest impact on local fishing communities and local tourism establishments. It has been estimated that the 27,000 km^2 of coral reefs and their associated seagrass beds in the Philippines, in their degraded condition in 1996, contributed a very conservative US$1.35 billion to the Philippine economy (White & Cruz-Trinidad, 2001). This figure includes values for fisheries, tourism and coastal protection analyzed in a similar manner to calculations by Cesar (1996) for Indonesian coral reefs.

In the last decade the coastal environmental issues perceived as exerting the most severe impact on the coastal and marine environment in Southeast Asia are given in Table 4.

After a decade the priority coastal environmental issues in the region remain basically the same even if the perception is carried over into the year 2020. An indication of this scenario resulted from a consultation with Philippine experts under the Global International Waters Assessment (GIWA) Project. Focused on five most important environmental concerns in Sulu-Celebes Seas, the result is shown in Table 5.

Table 4: Coastal environmental issues in Southeast Asia.

Issue	Immediate	Short-term	Long-term
Habitat destruction (*sm)	1	1	1
Sewage pollution (*sm)	2	2	3
Industrial pollution (*sm)	3	3	2
Fisheries overexploitation (*s)	4	4	6
Siltation/sedimentation (*s)	5	5	4
Oil pollution (*sm)	6	6	8
Hazardous waste	7	7	7
Agricultural pollution (*s)	8	8	5
Red tides (*s)	9	9	11
Coastal erosion (*sm)	10	10	10
Natural hazards (*sm)	11	12	12
Sea level rise	12	11	9

These issues are ranked in order of priority and classified into urgency categories, i.e. immediate, short-term or within the next five years, and long-term or within the next 10 years or more. Those with asterisks are identified with severe negative impacts on seagrasses (*s) or mangroves (*m) or both (*sm). Modified from UNEP (1990).

Table 5: Five priority coastal environmental issues in SE Asia, categorized on the basis of severity and rank.

Environmental concerns	2001		2020	
	Severity	Rank	Severity	Rank
Habitat and community modifications (*sm)	3	1	3	1
Unsustainable exploitation of fisheries and other living resources (*sm)	3	1	3	2
Pollution (*sm)	2	3	3	2
Freshwater shortage (*m)	2	4	3	4
Global change	1	5	1	5

Severity: 3, most severe impact, 1, slight impact; Rank: 1, highest, 5, lowest. Those with asterisks are identified with severe negative impacts on seagrasses (*s) or mangroves (*m) or both (*sm).

14.5. Worldwide Decline

Worldwide, there has been a rapidly increasing intensity of seagrass loss and decline (Thayer et al., 1975) and in many cases the magnitude of loss is high. In Asia-Pacific decline in seagrasses are well documented for 10 sites (25% of the total number of areas where declines have been reported (Short & Wyllie-Echeveria, 1996). These are confined in the following areas (Table 6).

In Southeast Asia, seagrasses are under threat from loss of mangroves which act as a "filter" for sediment from land, coastal development, urban expansion and bucket dredging for tin (Lean et al., 1990). Other impacts include, substrate disturbance, industrial and agricultural runoff, industrial wastes and sewage discharges. At the Seagrass Workshop held in Bangkok in December 1993, seagrass scientists of the ASEAN-Australia Living Coastal Resources (LCR) project have indicated that seagrass habitats in East Asia are rapidly being destroyed. In Indonesia about 30–40% of the seagrass beds have been lost in the last 50 years, with as much as 60% being destroyed around Java, while in Singapore, the patchy seagrass habitats have suffered severe damage largely through burial under landfill operations. In Thailand, losses of the beds amount to about 20–30%. Very little information on seagrass loss is available from Malaysia. In the Philippines, seagrass loss amounts to about 30–50%.

Table 6: Some countries and areas in Asia-Pacific with documented seagrass declines.

Country/region	**Area lost km^2 (%)**	**Cause(s)**	**Source**
Gulf of Carpentaria, Australia	150 (82%)	Cyclone	Poiner et al. (1989)
Hervey Bay, NE Australia	1,000	2 Floods + cyclone	Preen et al. (1995)
Botany Bay, Australia	Unknown	Dredging, explosion of sea urchin population	Larkum & West (1990)
Cockburn Sound, Australia	(80%)	Eutrophication from industrial development	Cambridge & McComb (1984)
Indonesia	(30–40%)	Siltation, pollution	Martosobroto (1994)
Philippines	(30–50%)	Siltation, eutrophication, unsustainable fishing	Fortes (1988)
Thailand	(20–30%)	Pollution, siltation	Sudara et al. (1994)
"World-wide"	12,000	(Unspecified)	Short & Wyllie-Echeverria (1996, 2000)

Table 7: Mangrove areas in Asia-Pacific and the margin threats.

Country	**Area (km^2)**	**Remarks**
Brunei Darussalam	70	Some portions lost to coastal development
Cambodia	100	Greater portion degraded via siltation
Indonesia	42,500	3,000 km^2 lost to coastal development
Malaysia	6,500	Vast areas lost to shrimp ponds
Philippines	1,320	Vast areas lost to shrimp/fishponds
Thailand	1,960	Vast areas lost to shrimp ponds
Vietnam	2,000	Decreased by about 45% since 1945
Pacific Islands	1,460	Vast areas lost to land clearing, agricultural development

Pacific Islands comprise Melanesia, Micronesia, Polynesia, Australia, and New Zealand.

In addition to the traditional uses of mangroves which, by and large, were fairly sustainable, recent population and economic pressures have led to an over-exploitation of the trees themselves as well as the conversions of the wetlands they occupy (ESCAP and ADB, 1995). Mangrove woods is being harvested for making charcoal (which in some cases is being exported); as a direct source of firewood; for the production of poles and other construction timbers; for the extraction of tannin used in the manufacture of inks, plastic and glue; and, for producing wood chips which are used as raw materials for the production of rayon (Aksornkoae, 1993). The vast area of mangrove forests in Asia-Pacific is partly shown in Table 7.

The rate of mangrove denudation in the Philippines is shown in Fig. 1. Thus, from 1918 to 1970, an average of 3,100 ha of mangroves were lost every year, increasing to about 8,200 ha annually from 1970 to 1988. The loss was due mainly to conversion to fishponds during the 1960s and 1970s. At present, 95% of the remaining mangroves are secondary growth and only 5% are old or primary which are mostly found in Palawan.

14.6. Monetary Value of Seagrass and Mangrove

The 27,000 km^2 of coral reef and seagrass ecosystems in the Philippines, equal to slightly more than 10% of the total land area of the country, are of significant value in terms of fisheries for food security, coastal protection, tourism, education, research and aesthetic value (Gomez et al., 1994; White, 1987; Courtney et al.,

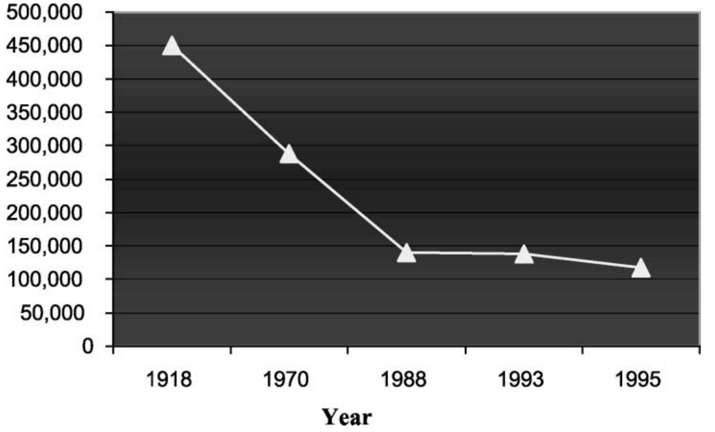

Figure 1: Mangrove denudation in the Philippines: 1918–1995.

1999). Most of the major commercial fisheries of ASEAN occur immediately adjacent to seagrass beds (Fortes et al., 1994). These beds are particularly important as nursery and feeding grounds for much of the sub-region's prawn and fish catch (Poovachiranon et al., 1994). In the Philippines, coral reefs with their associated seagrasses potentially could supply more than 20% of the fish catch (McManus, 1988). A total of 1,384 individuals and 55 species from 25 fish families were identified from five seagrass sites in the country. All members of these families have economic value mostly as food and aquarium specimens. Five times as many fish live over seagrass beds as over sea floors made up of mud, shells, and sand (Lean et al., 1990).

But there have been very few studies of the direct economic benefit humans derive from seagrasses. It has been estimated that the economic return from seagrass beds can be up to US$ 86,000 per acre (IUCN/UNEP, 1984). Based primarily on the fisheries they support, seagrass beds in Cairns, Australia, cost A$700,000 annually (Coles, 1986). Watson et al. (1993) found that the potential total annual yield from Cairns Harbor seagrasses for three major commercial prawn species was 178 t yr^{-1} with a landed value of US$12,325 annually. In Monroe County, Florida, the commercial fishery for five seagrass-dependent species was estimated at US$48.7 million yr^{-1}. Worldwide, recreational fisheries, diving and snorkeling are industries, which depend directly or indirectly upon healthy seagrasses beds (Heck, 2001). In an assessment of the economic value of the world's ecosystems, Coztanza et al. (1997) listed the value of the nutrient cycling function of seagrass beds at US$3.8 Trillion, the second highest among all the other ecosystem values listed. Estuaries ranks first, but again, seagrasses are often found in these habitats.

In the case of mangrove in Bacuit Bay, western Philippines, an economic analysis was made to examine the economic effects of sedimentation pollution on tourism and marine fisheries based on two development options: (1) to ban logging in the bay's watershed; and (2) to allow logging to continue as planned (Hodgson & Dixon, 1988). The results of the study are striking. The project estimated a reduction in gross revenue of more than US$40 million over a 10-year period with continued logging of the Bacuit Bay watershed as compared with gross revenue given implementation of a logging ban. The difference is due to projected losses from tourism and fisheries.

An estimated net annual economic value (in US$/ha) of Philippine mangrove areas for different levels of management is given in Table 8 (White & Cruz-Trinidad, 2001).

The total gain to the Philippines for protecting its remaining mangrove ecosystem is substantial. Using the conservative estimate of value from direct benefits of only US$600/ha/yr, the country gains at least US$83 million/yr in fish production and potential sustainable wood harvest from the existing 138,000 ha. If we could increase the area of healthy mangrove forest to 200,000 ha, the annual natural benefits would potentially increase to US$120 million for a gain of about US$37 million/yr.

In the Gulf of Fonseca in El Salvador, three different management strategies for mangroves were considered (Gammage et al., CSI Forum): partial conversion to semi-intensive shrimp farming and salt production; the do-nothing strategy of deforestation, land clearance and degradation; and the sustainable management option. A variety of different valuation techniques were used to assess the contribution of different products and services of the mangrove ecosystem. The sustainable management strategy enables more timber and fisheries benefits to be captured over a longer time frame than do the other management options.

Table 8: Estimated net annual economic value of wood and fish products from Philippine mangrove areas under different levels of management.

Level of management	**Wood products**	**Fish products**	**Total**
Mangrove plantation	156	538	694
Managed naturally regenerated	90	538	628
Unmanaged under stocked stands	42	538	580

Note: Wood harvest value based on average price of about US$12/cu m of wood; fish products based on average annual weight of fish and shrimp/ha associated with mangrove areas and an average price of US$0.80/kg; values based on Philippine Pesos, US$1 = PhP 25 in 1991.

Both the current management strategy and the partial conversion strategy yield net benefits of approximately US$7,500 per hectare whereas the sustainable management strategy generates a little over US$10,000 per hectare in ecosystem goods and services. The benefits from sustainable mangrove management can only be captured if existing patterns of resource use are modified. This requires fundamental changes in existing policy and legislation and in the institutions that administer and enforce these laws. Costanza et al. (1997) has provided new estimates of the services derived from mangrove ecosystems (e.g. shoreline protection, protection against rising seas, food and wood production, and habitat for wildlife) at US$10,000/ha/yr.

14.7. Impediments to Addressing the Issues

A review of seagrass literature produced over the past decade (1989–1997) showed a sustained increase in the scientific production in international journals. In addition the annual publication rate doubles every four years. Led by Western Europe, the Mediterranean, Caribbean and Australia, 33 countries have contributed to the knowledge base. Similar efforts in Northwest America and Southeast Asia are notably encouraging. However, scientists from only two countries produce half of the production, the focus of study on only 10% of the seagrass flora from only two biogeographic areas. In addition these studies are largely descriptive (>60% of papers), not synthetic, hence, with low predictive value useful to resource management.

Coordination in global seagrass research is extremely limited and fragmented, resulting in great uncertainties when scaling up the knowledge produced locally, so that broad-scale assessment of seagrasses is faced with great difficulty.

The sad state of research on seagrasses and mangroves is a reflection of the dismal state of marine science worldwide. In Asia-Pacific the latter has been confronted with the greatest barrier to its development and diffusion: the lack of effective linkages between science institutions (scientific production) and the productive sector (application). With it comes the other obstacles which, in the next century, would still be the shortage of funds for research, low salaries for staff, lack of access to needed technologies, weak technical support infrastructures, poor public appreciation of coastal resources and environment, and the relatively small number of researchers trained in promoting an integrated management approach. Unless there is a substantial change in the legislative agenda within the majority of developing Asia-Pacific countries especially in Southeast Asia, the lack of national commitments to support and encourage the development of marine science with focus on coastal resources management and protection will remain a major deterrent.

Until very recently efforts to manage the coastal and marine environment in the Philippines have focused mainly on identifying the problems and planning remedial and preventive measures. Past and increased awareness of the problems in the country have therefore not actually solved the problems that the marine and the coastal environment face. However, renewed attempts are being made at the national, regional and international level to address the problem effectively. On the other hand, countries of the region have joined various international and regional agreements to resolve the problem and yet huge tracts of seagrass beds and mangroves are increasingly being degraded and lost, and species depending upon them for survival, threatened. This is in part due to the fact that policy or decision makers are unaware of the features and values of these wetlands in their charge. While there is a large amount of knowledge on the threats to the habitats, this has not been placed in context, making it difficult to use to assess the state of global seagrass and mangrove resources or to establish priorities for their management With increasing pressures from population growth, the rising number of megacities, accelerated economic development, depletion of coastal resources, degrading of coastal water quality and increasing resource use conflicts, there is a critical and urgent need to move from information gathering and planning to management solutions of coastal and marine environmental problems.

In addition to the inadequacy and ineffectiveness of efforts to conserve and manage seagrass and mangrove habitats, functional coordination among the stakeholders is nowhere in sight. Depicted in Fig. 2 below, this is the biggest impediment to success in coastal habitat protection.

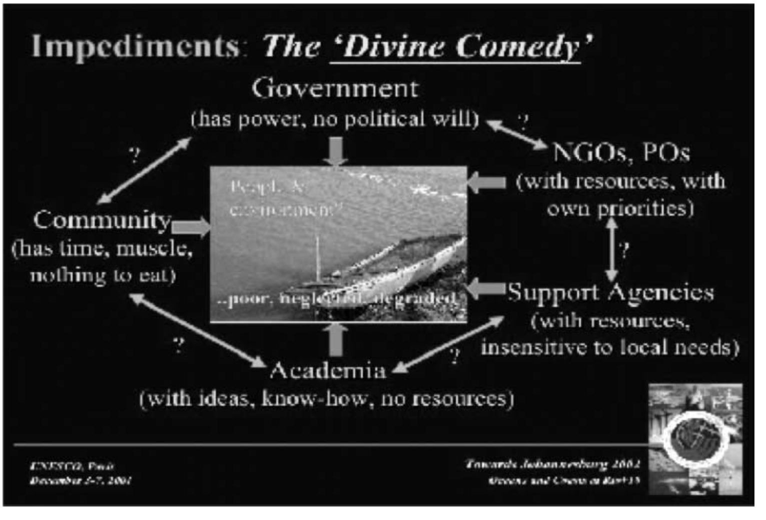

Figure 2: Lack of political will and non-coordination among agencies: the biggest impediments to success in coastal habitat protection.

14.8. Conservation and Management Strategies

This section of the chapter contains some recommendations or steps to be undertaken in order to arrest coastal habitat destruction in the Philippines. They are given under two categories: those specific for seagrasses and mangrove habitats and those for the protection and management of the country's coasts as a whole. Central to the formulation and implementation of these recommendations are conditions, which should be adhered to in order to ensure positive feedback from the people. These include: adherence to rules and regulations; participatory in character; built on consensus; incorporates capacity building and institutional strengthening; gender sensitive, and with regional and global perspective.

14.8.1. For the Management of Seagrass and Mangrove Resources

Focus Research on Priority Management Issues. In the last 5 years the Philippines (with Thailand and Vietnam) has seen remarkable efforts in understanding seagrasses and mangroves and utilizing research data and information to bring about a positive change in approaches to reverse the degradation and loss of these habitats. Funding agencies like the European Union have become more receptive to the clamor to understand the responses of these plant communities to siltation derived from deforestation (Project TS3*-CT94-0301 of the STD-3 programme of the European Commission). The percent sediment yield from Southeast Asian rivers to the ocean is the largest on earth (Milliman & Meade, 1983), hence, siltation remains the priority coastal environmental problem in the region. The significant output of the project has been a basis for another, more focused research initiative to predict their resilience and recovery given Southeast Asia's coastal environmental conditions (EU Project INCO-DC Contract No. ERBIC18CT980292). In many countries in the region, mangrove reforestation and afforestation have been enhanced to improve the productivity and protective capacity of coasts. The increased awareness on the important role seagrasses and mangroves play in the conservation of coastal biodiversity have prompted NGOs to contribute significant efforts and resources in understanding these plant communities and protecting them to conserve endangered species like dugongs and turtles. The documented loss of seagrass habitats has prompted concern among managers and scientists to search for indicators of seagrass ecosystem health. In the Philippines experimental approaches, based on the work of Fonseca et al. (1982), have been revitalized to investigate the potential of seagrass (natural and artificial) to restore or rehabilitate degraded coastal areas (e.g. mine tailings disposal sites).

Remote sensing has been used in the region on a pilot project basis to obtain data on suspended sediments in the water column, topography, bathymetry, sea state, water color, chlorophyll-a, sea-surface temperature, fisheries, oil slicks, and submerged and emergent vegetation, including mangroves and seagrass meadows (Kam et al., 1992). However, despite its undoubted potential, remote sensing has some severe limitations and practical problems when applied to the coastal zone in particular, which arises primarily from tidal water level fluctuation which influences the level of penetration. A further constraint to the use of remote sensing, particularly the use of satellite imagery, in the coastal zone of the tropics and sub-tropics are their propensity for cloud cover, which hinders the taking of images. In spite of these difficulties, remote sensing is an appropriate method for addressing information needs in coastal decision-making and it is therefore surprising that in the Asian and Pacific Region, remote sensing has not been integrated into national coastal development processes (Kam et al., 1992).

Complementing remotely sensed images of the coasts, there is an urgent need for basic and directed research the results of which are contained within an integrated information management system that can be used to monitor the rate of change of Philippine wetland resources, and a need to make it available for planners of seagrass and mangrove conservation.

There is a dearth of data on the current status of the seagrasses and mangroves in the Philippines: their actual overall coverage, density, growth patterns, their responses to perturbations, and use patterns. Similarly, almost no data exist on fuel wood and timber requirements, siltation, pollution, and chemical runoff into rivers and water bodies that drain into the mangroves. The data that exist are scattered and inconclusive and do not provide sufficient detail for the development of parameters to guide and monitor the sustainable extraction of seagrass and mangrove resources. In order to work towards more sustainable seagrass and mangrove management in the Philippines, key gaps in data collection need to be addressed. Data need to be collected on key biological and human-environment indicators that will guide policy and set parameters for sustainable resource use. The regions in the world where large-scale seagrass declines have been recorded should convince international and national governments of these regions to focus more integrated research efforts on these critical areas.

Interestingly, Fig. 3 gives a picture, in terms of publications, of the progress made in seagrass and mangrove research in Southeast Asia from 1985 to 2001. In part it also reflects the available expertise on seagrasses in the region. It can be seen that while the trends in the studies remained basically similar, those in mangroves have been undertaken with more vigor than seagrasses. This is primarily due to the relatively young state of seagrass science in most parts of the region. One reason for the slump in the production of published works from 1985 to the 1990s had been associated with the dramatic decline in the economies in

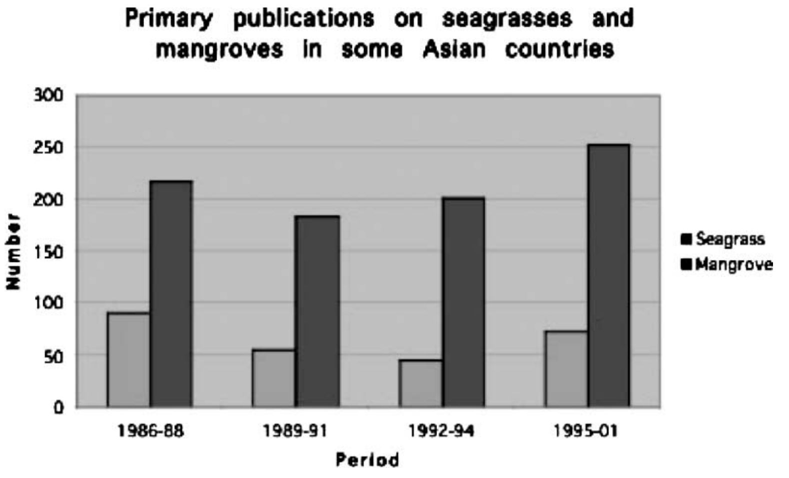

Figure 3: Publications on seagrass and mangroves in Southeast Asia from 1985 to 2001.

many parts of the region. This brought about the increased involvement of scientists in technical services (consultancies) where their outputs were merely published in grey literature. The renewed interest in quality outputs in 1994 reflects the impetus both academic institutions and funding agencies placed on marine science despite a continued decline in economic conditions.

Develop an Integrated Framework for Action: Putting Our Acts Together. For seagrasses the International Seagrass Biology Workshop series have produced the essential elements in conserving and sustainably utilizing the seagrass resources of the world. These elements consists of: (1) the needed linkages among seagrass scientists and practitioners from all parts of the world; (2) the mechanisms to ensure access and transmission of data and information; (3) sustained research activities on the dynamics of the ecosystem; and (4) modest support from academic and funding institutions. With support from UNESCO–IOC the association initiated the establishment of the World Seagrass Association (WSA). With the help of the French and Italian Governments, the association now serves as the clearinghouse of all activities on seagrasses in the world. At the time of its establishment, the Seagrass Monitoring Project (*Seagrass* Mon) and Seagrass Network (*Seagrass* Net) were developed and operationalized to implement the plan of action for seagrasses. Approved at the third International Seagrass Biology Workshop in the Philippines (1998), the *Charter for Seagrasses* was adopted, laying the principles that guide research and development of seagrasses in the world.

In the case of mangroves, a coalition of community groups, researchers, non-governmental, private sector, and governmental agencies in Honduras and El Salvador has developed a platform for action for the sustainable management of the mangroves in the Gulf of Fonseca (Gammage et al., http://www.unesco.org/csi/wise/fonseca1.htm). The platform for action has been developed as an advocacy tool for activists, NGOs and community groups to allow these organizations to engage in dialogue with both the government and the private sector. It represents the outcome of more than eight years of collaborative activities to explore the competing interests for mangrove resources of community groups, aquaculturists, farmers, salt producers and fisher people. The platform advocates for legislative, institutional and procedural changes to be set in motion to begin to harmonize the diverse interests of these multiple stakeholders in the ecosystem. The recommendations provide guidelines for a process that must be set in motion if these unique resources are to be preserved. It is essential that policies and programmes are devised that can simultaneously meet development goals and guarantee the health and well being of the ecosystem. Without such efforts, the mangroves will be degraded and a wealth of resources that they secure will be lost.

As in the United States a coastal initiative tasked to codify information needs in different regions should be developed. A knowledge base can then be used to formulate local, regional, and national conservation strategies for seagrasses and mangroves that are biologically and ecologically acceptable and economically sustainable. The goal of these strategies should be net enhancement of natural capital for the sustainable use by present and future generations. These strategies should include mechanisms for managing and protecting the ecosystems sustainably in the face of global change; they should also employ the best, most up-to-date scientific information available, and should evolve to incorporate new information as it is generated. A periodic review will have to be designed to answer among others, the following questions: Is the scientific information being used actually relevant to the policies and decisions that must be made? Has information been provided in a way that facilitates its use? Is the information timely? Is it credible? Do decision makers understand it? Do stakeholders understand it?

The initiatives above are currently seeking more support to continue their worthy scientific and advocacy campaign and hope to develop a series of popular education materials, radio and television advertisements that address the concerns raised in the platform for action and provide targeted information to a range of parliamentarians and municipal officials in coastal districts.

Undertake an Economic Valuation of the Resources and of Relevant Policy Changes. In the past many environmental problems were regarded as local and straightforward. They were seen as being easily regulated by elementary command-and-control instruments. Now, more environmental problems have come to be recognized as having greater complexity than previously thought, with wider impacts than first-round or local effects. This gave rise to market-based approaches to environmental regulation.

As mentioned above seagrasses and mangroves provide environmental goods and services. In most parts of the developing countries of Asia-Pacific, however, these are being used unsustainably without regard to the external costs that their actions impose upon the ecosystems and upon others who also depend upon them. The "total economic value" of seagrass and mangrove ecosystems should be estimated using a cost-benefit analysis to compare the sustainable management of the habitats with alternative use scenarios.

Market mechanisms will only be successful if they reflect the preferences of citizens as individuals, both nationally and internationally. Environmental economics has made considerable progress over the last half century in devising methods that attempt to quantify the strength of preferences for various environmental amenities, and to identify and define the extent of the market (people) affected by environmental changes. Environmental valuation of the resources and the benefits of policy change relative to these resources are thus extremely important. It should be emphasized, however, that in the process we face the dilemma of pricing the priceless, of quantifying the unquantifiable, of creating common standards for things apparently unequatable (de Groot, 1992). Fonseca (personal communication) argued that trying to determine the monetary value of an obviously rich and biologically diverse resource as a seagrass ecosystem might be a waste of time, for this will only further delay its development. But until better instruments and methodologies are found, giving money values to ecosystem functions may help convince policy makers and financiers of development projects of the importance of nature conservation and the true meaning of environmentally sustainable economic development. In the valuation process, however, ecologists should be involved more actively with the view that the whole exercise is purely for the purpose of management. This is because if they are not, others who are less informed of the true worth of the environment eventually will, and attach to it a much lower price. The low values attached to coastal resources are one principal reason for their continued loss.

Forge Public–Private Partnerships to Conserve and Manage Seagrass and Mangrove. In order to conserve and manage seagrass and mangrove resources sustainably, it is necessary to use the relevant scientific information that is currently available. It should inform conservation strategies at the local, regional, and national levels. It is also necessary to generate new knowledge to fill in gaps in our understanding of the ecosystems in the face of environmental change. Hence, we should start by using the knowledge that we do have, organizing it electronically, and providing it to all parties that need it. To accomplish this, we will need to form partnerships among governmental organizations at international, regional, national and local levels, and between them and the private sector. These partnerships, using up-to-date information, can begin the process of developing coordinated strategies by designing best management practices and further sharing information.

The Workshop, *Seagrass-Watch: Community, Coasts and Clean Seas* held in Hervey Bay, Brisbane (12–15 October 2001), is a commendable effort on the part of Queensland Department of Primary Industry to increase people's awareness on the importance of seagrass resources. The workshop discussed various approaches to involving community groups in seagrass monitoring and research and/or the wider question of the role of communities in science and in decision-making.

Ensure a Functional Coordination Among Concerned Agencies. Coordination of actions among various agencies mandated to protect the coasts would help to eliminate duplication of effort and therefore save funds that could be invested more wisely. Coordination also would illuminate research areas in which agencies and academia could cooperate, and would facilitate the development of information systems that would serve not only management agencies but also the public. The coordination process should provide forums for discussion, so that lessons learned by one entity can be instructive to many. At present, the region is probably not gaining the full value of lessons learned from policy successes and failures. Forums also provide an avenue for input from the public and from the private sector, which in itself can be of great value in time and expense saved, opportunities for understanding gained, and in litigation avoided.

The absence of coordinated strategies for conservation is one factor that allows the continued degradation of the region's natural coastal capital. If coordination of management and research activities is not achieved, many of these agencies will continue to manage inefficiently or to work at cross purposes with each other. This in turn leads to unnecessary expenditures, interagency conflict, public dissatisfaction, and mismanaged natural resources. In the absence of coherent strategies, it will become more and more difficult to bring the results of up-to-date research into management and policy decisions.

Increase the Content of Seagrass–Mangrove Information Repositories. At present the amount of information that any repository of knowledge on seagrass and mangrove ecosystems in the Philippines and Asia-Pacific can provide does not reflect even a small percentage of the body of ecological and other biological knowledge. There is much information available in the scientific literature and even in databases that is not part of any of these structures and is not readily accessible, but which could be extremely useful in the generation of habitat conservation plans and other ecosystem management strategies. Steps should be taken to increase the online electronic information content of these data repositories via the allocation of a certain percentage of all research funding specifically for the long-term management of the data and information generated.

Adopt "Wise" Management Options. True management of seagrasses resources is in its infancy in the Philippines. On the other hand, years of experience characterize local efforts in conserving and managing mangrove resources in the country. These point to four options, which have the highest probability towards a successful management of the habitat. These are:

1. Mangrove nursery establishment and management-site selection, design, operation and management of nurseries for the Philippine mangrove species. Nursery technologies ensure the availability of planting materials and the production of high quality seedlings.
2. Mangrove plantation establishment and management-developing and managing mangrove plantations and the remaining natural forest stand to maximize the benefit to the coastal ecosystem; non-regulatory techniques (training and education, research and monitoring) are especially relevant strategies in mangrove plantation planning.
3. Community-based forest management agreement (CBFMA)-a production sharing agreement entered into between a community and the government to develop, utilize, manage, and conserve a specific portion of the forestland, consistent with the principles of sustainable development pursuant to a community Resource Management Framework (CRMF). The latter is a document that defines the terms and procedures for accessing, using and protecting natural resources within the CBFMA.
4. Fishpond restoration-modifying abandoned or illegal fishponds in CBFMA areas to harvest firewood, poles, shells, fish, crabs and to provide food and shelter to crabs, shrimp, shells and fish in coastal waters. Aquasilviculture, which is the conversion of a fishpond area into a site where mangroves can grow and fish can thrive, is suggested as a fishpond restoration strategy.

14.8.2. For the Protection of the Larger Coastal and Marine Environment

Localize Sustainable Development Through Sound Governance. Given the largely similar and specific operational and field conditions among the coastal communities in the Philippines and the difficulty in translating experiences from other parts, more effort should be invested to encourage the exchange of experiences and successful examples and models among these states and regions.

However, this initiative should not detract from the value of cooperation and twinning with countries from other parts of the world.

Management programmes for coastal regions and small islands must be based on the interests of all stakeholders, and should not be exclusively top-down or bottom-up orientated. Universities, NGOs and agencies must work together to develop an action research agenda, which supports sustainable income-generating activities.

The issues confronting the coastal and marine environment can be effectively addressed by adopting certain innovative measures, which start with general recommended strategies and thereafter focusing on certain specific objectives that uphold these strategies. The latter include: crystallizing the people's vision for coastal marine ecosystem development; providing and exercising political will, dynamic leadership and the courage and determination to pursue this vision; and getting the people, government, and all sectors (private plus donors) to support this vision. The major objectives are envisioned to guide the environment sector in the performance of its evolving role in leading the implementation and enforcement of laws and regulations.

In the Philippines, the environment sector of the government is perceived as an "unfriendly" entity whose personnel are the first to violate the laws they are mandated to uphold. Its policy programs are heavy on addressing the "victims," not the "culprits." This is one major reason for the general distrust and non-confidence people have on the government. What people wants now is for the sector to work by example, demonstrating with honesty and a low profile, what it can do and cannot do, its successes and failures. The latter has been difficult because in most cases, activities have been practically monopolized by the sector, not involving the people in the process. Many that see it if only the sector could effectively enforce the laws, at the same time help educate the people; at least half of the problems would be solved.

Ensure High Quality Scientific Publication: A Shift from Description to Synthesis. The problems of insufficient information arising from the low priority nations in the region accord marine research and the poor quality, largely descriptive, data available are reflected in the share Third World countries have in the so-called international scientific literature. Although developing countries encompass 24.1% of the world's scientists and 5.3% of its research spending, most leading journals publish far smaller proportions of articles by authors from these regions (Gibbs, 1995). This is shown in Table 9.

This near invisibility of less developed nations in international scientific literature may reflect the economics and biases of science publishing as much as the actual reality of Third World research. Such invisibility to which mainstream science publishing condemns Third World research, however, thwarts the efforts

Table 9: Share of mainstream journal articles.

Country	% of total for all nations
USA	30.817
Japan	8.244
UK	7.924
France	5.653
Canada	4.302
Taiwan	0.805
Hong Kong	0.205
Singapore	0.197
Thailand	0.086
Malaysia	0.064
Philippines	0.035
Indonesia	0.012
Cambodia, Laos, Vietnam	0.006

Only the top five countries and 10 from Southeast Asia are shown (after Gibbs, 1995). Data are taken from papers published in 1994 by some 3,300 journals included in the Science Citation Index, a commercial database.

of poor countries to strengthen their indigenous science journals -and with them the quality of research in regions that need it most. It may also deprive the industrial world of critical knowledge. As Christopher T. Zielinski of the World Health Organization puts it, "The 2% participation in international scientific discourse allowed by Western indexing services is simply too little to account for the scientific output of 80% of the world."

In Ecology, *Trends in Ecology and Evolution* is a top reputable journal. But in 1994, it accepted no article for publication by authors from any of 100 developing countries. The low representation accurately reflects the poor quality of science in poor countries: "Environmental Science in developing countries is indeed lagging behind the rest of the world, just as you would expect," says, William H. Glaze, editor of Environmental Science and Technology. "Not only is it old-fashioned, but sometimes it is just not very well done. The documentation is poor, and the experimentation does not meet our standards."

In coastal zone management this "trend" is similarly reflected. There are indications of use of generally poor quality, largely descriptive, hence, with low reliability information (represented by "gray" literature). This is demonstrated in Table 10 (after Lacanilao, 1995).

Table 10: Publications in Coastal Zone Management with "good"- and "poor"-quality information.

	Total articles cited	**No. from refereed journals**[a]
MPB 26:540, 1993. Environmental impact Assessment — a review of its aims and recent developments	36	30
East-West Centre SPREP Training Manual, 1989. How to assess environmental impacts on tropical islands and coastal areas	102	4

[a] Journals covered by indexes of the Institute for Scientific Information

One implication is that high-quality reliable ecological knowledge (quantitative studies, published in refereed journals) is not or only peripherally utilized as guides in the management process. It also implies our inability to access valuable information much needed in coastal zone management. For example, a major reason why marine conservation lags behind terrestrial conservation is our ignorance of the sea's vulnerability to us, which we can only understand largely through ecological research. We know less than we need to, and the little knowledge that is available of traditional users of the sea and marine ecologists is not available to all who need it. This is the reason why we lack broadly applicable marine ecological theory. Most countries in the region lack even basic background information of currents, species inventories, and especially ecosystem dynamics that are fundamental to informed decision-making. This lack of knowledge prevents coastal managers from using a simple set of standards to guide all their decisions. Because so many decisions are based on incomplete information, how decision-makers rule when there is insufficient knowledge is a central question in coastal zone management. Its answer will determine whether sustainable use becomes a living, working reality or a fondly disregarded concept having no relevance to the world in which we live.

Position Science and Technology Centrally in Economic Development Policy. Along with changing regional and economic structures, the Philippines sought to position science and technology (S and T) more centrally in its programs for economic development. Production and trade in the western Pacific Rim countries in the latter part of the decade reflect a regional economic identity with science at the core of each development strategy. There is a general drive to orient science in the public sector towards the marketplace and to look for future S and T growth and application within the business sector.

There are considerable variations in the level of industrial development among Asia–Pacific nations. More importantly, this condition is invariably correlated with the status of S and T links between sectors. Science across the region is now more context-oriented and focused on the research problems articulated by the private sector. Driven both by the growth in the business and university sectors and by government policy, alliances between scientific institutions and business enterprises have multiplied. A recent Asia Pacific Economic Cooperation (APEC) study identified keener competitiveness, greater complexity of knowledge, the pace of technological advances, and increasing financial pressures on universities as being among the major factors combining to drive sustained growth in university-industry collaboration.

Most countries of the region have recorded a rise in gross domestic expenditure on R and D (GERD) in the 1990s. This is shown in Table 11. In absolute terms, GERD has increased steadily across the region, but because of the rapid and sustained growth of GDP in economies like Singapore, the ratio of GERD to GDP has not always improved. In the case of China, it has actually

Table 11: GERD in Selected Pacific Rim Economies as a Percentage of GDP Germany and the USA are given for comparative purposes.

	1981	**1991**	**1995**
Australia	1.0	1.3	1.6
China	0.8	0.7	0.5
Indonesia	–	0.2	0.3
Japan	2.3	3.0	3.0
Republic of Korea	0.6	1.9	2.4
Malaysia	–	0.8	0.4
New Zealand	1.0	0.9	1.0
Philippines	–	0.2	0.2
Singapore	0.3	1.3	1.2
Chinese Taipei	0.9	1.7	–
Thailand	0.02	0.2	–
Germany	2.4	2.7	2.3
USA	2.4	2.8	2.5

Source: S and T Analysis Section, Department of Industry, Science and Technology, Australia (1996), based on OECD and national data.

fallen. Notwithstanding the fact that accelerated economic growth has sometimes masked increases in real R and D expenditure, recent estimates suggest that by 2005 the East Asian economies alone will spend more on R and D than the USA (Table 11).

The Medium Term Programme for S and T in the ASEAN for 1996–2000 focuses on developing a technology information network to link institutions and existing networks and thereby promote information sharing, human resource development, and technology transfer in the rapidly developing technology-intensive business environment. A first step in this direction was the establishment in 1996 of the ASEAN Science and Technology Management Information System (ASTMIS).

With respect to the small Pacific Island countries, what they have in common is that their economic growth has been stagnant to low since the 1980s, that their indigenous communities are largely subsistence-based. However, most have prospects for significant industrialization. Scientific research has been mainly a public sector activity concentrating on natural resource utilization. Stagnant economic growth, alongside high population growth and the resultant boom in unemployed youth, has sharpened awareness on the need for alternative development paths. Here, scientists have increasingly focused on meeting people's needs through participatory R and D, spearheaded by several NGOs. This necessitates examining what people know and do, including investigating traditional knowledge and technologies.

Adopt the Integrated Coastal Area Management Philosophy. Integrated Coastal Management (ICM) is "…a process that unites government and community, science and management, sectoral and public interests in preparing and implementing an integrated plan for the protection and development of coastal ecosystems and resources." The overall goal of ICM is to improve the quality of life of human communities who depend on coastal resources while maintaining the biological diversity and productivity of coastal ecosystems.

Ecological theory as described in standard textbooks on ecology, is seldom applied directly to coastal zone management in Southeast Asia. But ecological knowledge — including not only theory, but also facts, observations, research results, observations, syntheses, models, and methods of investigation — has been extremely important in developing approaches to a wide range of environmental problems. This stems from the "man-environment" model, given below, which identifies the essential and crucial role of ecology:

ENVIRONMENT	**《ECOLOGY》**	**MAN**
natural processes and components	**《functional interrelations》**	**human needs and activities**

From the model, ecology is the key to a sustainable use of the environment and its resources. This is because it investigates the nature of the linkages inherent in or resulting from the use of these resources by humans, defining limits (carrying capacity). Thus, it provides the information and a means by which these relationships could be understood so that the necessary actions could be implemented.

But ecology alone is not sufficient to address effectively coastal zone management issues. No matter how much biologists know about the population dynamics of the sea cow or the ecosystem dynamics of seagrass beds, it will not be possible to protect or use them sustainably unless we understand the human causes and consequences of their increasing rarity. Having better information about populations, species, and ecosystems is essential, but not sufficient: Decision-makers also need much better information about human causes and consequences of protecting and using living resources. Our success or failure is rooted in our cultures and economic activities. These can be forces for conservation and sustainable use, or they can be forces that eliminate species and ecosystems. For life in the sea, the diversity of human cultures offers both promise and risk, but in Southeast Asia, the promise outweighs the risk. Indeed, coastal zone management issues are deeply rooted in society and culture, which require, for their resolution, significant input from ecology. It is becoming more acceptable that, as Salwasses (1993) puts it, "…long term management of resources must be adaptive rather than deterministic. And it must be economic and political rather than scientific."

These understanding and implementation are realized only when the importance of nature and a healthy natural environment to human welfare is fully reflected in economic planning and decision-making, i.e., when ecological data are translated into useful information for planners and decision-makers. What is most lacking is a simple but effective method for local planners and decision-makers to decide on the best alternative use of a particular natural area, including the option to conserve it in its natural state (de Groot, 1992).

Hence, the current problem facing Philippine decision-makers is how to manage the apparently conflicting activities and uses of the coastal zone and its marine environment. Coastal resources are impacted directly by activities in the zone, and activities well inland, which are transmitted by rivers or carried by currents by other regions and countries. Therefore, the management of coastal zone requires a multi- and inter-sectoral approach. Control and management of the living resources of seagrass beds and mangrove forests, and the sediment areas between these systems must be a multi-sectoral responsibility, involving many government and private sectors at all levels. Here, the decision-making process must rely on science and technology, and make hard decisions for the long-term management of human uses of the coastal zone — decisions that will put to test the democratic process as some human activities will have to be curtailed.

14.9. The Challenges

Today's problems are a result of successes as they are defined in yesterday's terms. The Philippines needs good men and women who can make good plans. The planning process must start with a value discussion that ends up with general and operational goals: what kind of coastal development does the Philippines want and what kind of social, cultural and environmental qualities does its people want to keep or strive for? Because of great variety in culture and interests in the country, such goals should be decided after a comprehensive planning process with broad input from all interest groups. These comprehensive plans need to be tested for realism under conditions of limited resources and established environmental quality requirements. Recognizing that the regenerative capacity of the country is limited, as well as its resource base, a strong motivation exists for the use of carrying capacity philosophy as the basis for national planning. These relatively scarce resources must be managed in the context of competing demands, and the environment must be considered as the region's inhabitants change their social, technical, or economic activities.

Here, natural and social scientists and engineers have a social obligation to seek a solution. They should develop a stronger sense of ethical responsibility. Science and engineering are clearly necessary to allow us to use the world efficiently and to form a stable relationship with it, but they are not sufficient. Unless humanity addresses effectively the issues on population, excess consumption, inappropriate technology, and cultural insensitivity, science and engineering will not be able to help our ailing world.

In the longer term, sound management of coastal and marine ecosystems would depend on an educated community in which members understand the importance of a mix of conservation, development and community participation. Past, and probably present governments have not tried to educate the population towards a more realistic way of life, nor to convince them that because of globalization, for example, the world is shrinking, that modernization needs hard work, and that we are obliged to support the sustainability of our environment.

The science community needs to develop and nurture an ethic that views the seas as a resource in need of our stewardship and not simply a commodity. The extent to which local community participation in marine environmental protection and resource management can be fostered will be a significant factor in determining the quality of the marine environment and the availability of its resources in the future. Indeed, people of the Philippines have been "biting

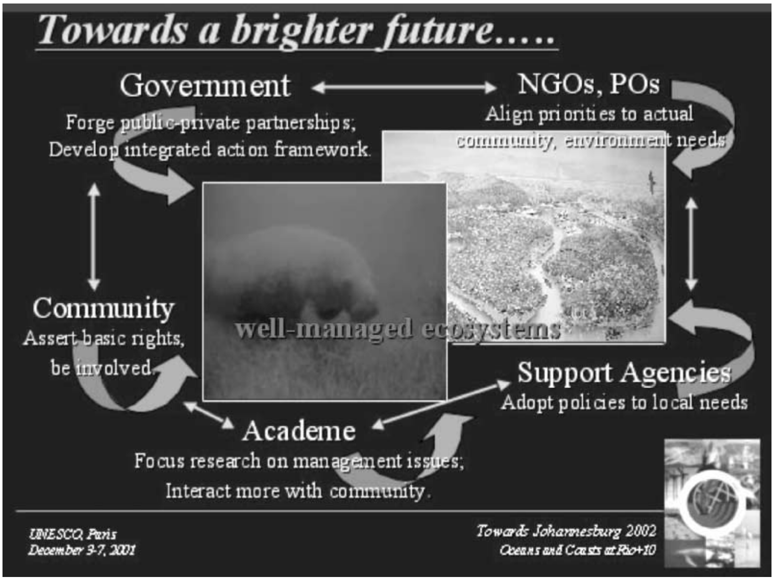

Figure 4: The “ways forward” in conserving and managing seagrass and mangrove in the Philippines (see Fig. 2 for related information).

the hand that has been feeding them for generations.” In so doing they have been slowly foreclosing options for the future.

In relation to the role of stakeholders in the conservation and management process, Fig. 4 shows the ways forward in terms of what they can do to remove the impediments depicted in Fig. 2.

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Chapter 15

Economic Valuation of Mangroves for Improved Usage and Management in Thailand

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Abstract. The role of mangrove ecosystem and its importance for management is examined. A dynamic simulation model to assess the cost and benefit of mangrove reforestation is developed and applied to a site in Thailand to test the applicability of the model. The model contains three main components: the natural mangrove ecosystem function, the interaction of human economic activity on the mangrove ecosystem and the economic model for the optimal utilization of the mangrove. The application of the model to the Bandon Bay area in Suratthani Province, southern Thailand, is discussed.

15.1. Introduction

This chapter looks at how valuation of mangrove ecosystem can be used to assess the cost and benefit of mangrove reforestation. The ecological role of mangroves in the coastal ecosystem is briefly reviewed. The economic evaluation of these functions as reported in the literature is reviewed in Section 15.2. A model, written with STELLA, is proposed in Section 15.3, with reference to a case study located in Bandon Bay, Suratthani Province, in Thailand. The simulation results with the model are reported in Section 15.4. Section 15.5 discusses the results and concludes with direction for future work.

15.1.1. The Ecological Role of Mangroves in Coastal Ecosystems

Mangroves are plants that grow in the tidal area between fresh and seawaters. Identification of plant species and explanation of how these plants can survive in the particular conditions of the tidal region show a high degree of adaptation of the plant community.

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Box 1

- *regulation functions* — protection against harmful cosmic influences; protection of the local and global energy balance; regulation of the chemical composition of the atmosphere; regulation of the chemical composition of the oceans; regulation of the local and global climate; regulation of runoff and flood prevention; water catchment and groundwater recharge; prevention of soil erosion and sediment control; formation of topsoil and maintenance of soil fertility; fixation of solar energy and biomass production; storage and recycling of organic matter; storage and recycling of nutrients; storage and recycling of human waste; regulation of biological control mechanisms; maintenance of migration and nursery habitats; and maintenance of biological (and genetic) diversity.
- *carrier functions* — human habitation and (indigenous) settlements; cultivation; energy conversion; recreation and tourism; and nature protection.
- *production functions* — oxygen; water; food and nutritious drinks; genetic resources; medicinal resources; raw materials for clothing and household fabrics; raw materials for building, construction, and industrial use; biochemicals (other than fuel and medicines); fuel and energy; fodder (animal feed) and fertilizer; and ornamental resources.
- *information functions* — providing esthetic information; providing spiritual and religious information; providing historic information; providing cultural and artistic inspiration; and providing scientific and educational information.

de Groot R. S. (1992). Functions of Nature: in Environmental Planning, Management and Decision-Making. Wolters-Noordhoff BV, Groningen, The Netherlands.

Several functions have been identified for mangroves, ranging from the ecological to the human uses, according to de Groot's classification, as shown in Box 1. Of particular interest are the ecological roles such as nutrient recycling, the nursery functions, in addition to the direct use of mangroves as a timber source and for harvesting fishery products on-site. The ecological roles are often externalized to the off-site area outside the mangrove, and therefore liable to be ignored when a decision is taken to convert mangroves to other uses. In particular, mangroves in Thailand have been converted to other uses, such as shrimp farms, construction sites or other non-mangrove uses. However, awareness of the external impacts of loss of mangrove areas has increased in recent times, and there is now an active movement to replant mangroves in many areas along the country's coastline. What is the cost and benefit of such mangrove reforestation activities? To address this question, it is necessary to be able to quantify the benefits, external and internal, of mangrove reforestation.

15.2. Problem Statement

The problem of economic evaluation of mangrove for management is one of addressing the many ecological functions of mangrove in a framework of cost–benefit analysis. How much would an increase of 1 m^2 of mangrove contribute to human welfare and hence would justify the cost?

15.3. Review of the Literature

Many attempts have been made in the literature to address the economic valuation issue. A good review is Spaninks & van Beukering (1997) (S&B). The study examines various mangrove functions and methods to assess their economic values, and finds that "most studies limit valuation to use values: the availability of market prices or market prices for substitutes means that the valuation of use values is relatively easy."

On indirect use values, based on ecological functions, the study finds the following:

"Most studies limit indirect use values to the nursery function." As stated in the chapter, "the value of this function… depends heavily on the ecological linkage between mangrove area and fish stocks." The chapter comments "in most cases, valuation of the impacts of a management alternative on catches in off-site fisheries is based on somewhat arbitrary assumptions, rather than on detailed scientific information."

In addition, studies also "ignore price changes and other economic reactions, and use simple multiplication by market price to value catches."

As noted by S&B, access conditions also are important in determining catches, as shown by Freeman's extension of the Ellis and Fisher model with a production function. This has been demonstrated in Aniyar (2002). Investigating the effect of a decline in the mangrove area by 50% from an equilibrium situation, under open access, there is no change in the rent of fisherman, whereas under regulated access, a change in the mangrove area will reduce rent, or to keep rent constant, the number of fishermen must be reduced. Thus, it is shown that the economic value of mangrove depends not only on the ecological linkage between mangrove and fish catch, but also the effort used for obtaining the catch, as well as the access condition which determines the level of such effort.

The examination of the mangrove–fishery linkage has been carried out with a variety of approaches, as already noted in our summary of S&B. In their own study, S&B takes the approach of estimating a "maximum allowable sustainable catch per year" based on cohort analysis of juveniles and an assumed optimal exploitation rate, to obtain the value of fish catch for the analysis. Different management regimes produce different nutrient productivity of the area, which translates to fish productivity. Unfortunately, in their paper, no specific mathematical expressions are given for the derivation.

In Aniyar, the approach of using a dynamic simulation model is adopted. The different relationships are modeled with the STELLA program, which allows for a dynamic simulation of the model until equilibrium values are reached. However, the parameter values used are assumed, based on "hypothetical (but well informed) interactions between mangrove forest, a single species of fish and fishery activities. The carrying capacity is assumed to be proportional to mangrove biomass."

15.3.1. On Bandon Bay, Suratthani

The Bandon Bay area of Suratthani province lies to the west of the Gulf of Thailand. The area receives the inflow of freshwater from the Tapi river. The estuary area is fringed with mangrove forest, which has been reduced by conversion to shrimp farming and urban development. In the bay area, mariculture is practiced, particularly the farming of oysters, mussels and clams.

The nutrient status of the bay depends on the inflow of the nutrients with the natural river flows, as well as the human-induced nutrient discharge from the upstream urban settlements and industrial developments, as well as farming activities including shrimp farms. The conversion of mangrove to shrimp farms also has a bearing on the amount of nutrients entering the bay. A number of studies were carried out under the Thailand country study component, reported in the SARC/WOTRO/LOICZ 2001 study and Suthawan Sathirathai 1998 study for EEPSEA. The studies show that: (1) the nutrient status of the bay area is affected significantly by human activities, (2) the change in the nutrient status affects the biological productivity of the bay area and (3) shrimp farming imposes an economic cost in terms of loss of biological productivity which can be approximated by the foregone values of oyster production in the bay area.

Suthawan's case study of a village located in the mangrove area shows that there is significant direct use value of the mangrove forest, and that the economic value of the mangrove–fishery linkage can be estimated with the Ellis–Fisher–Freeman (EFF) model depending on the assumed demand elasticity and the cost function.

However, the above-mentioned studies are not integrated into a single quantitatively defined model, so the conclusions derived are at best qualitative, partial, and in terms of time sequence, ambiguous.

This chapter attempts to bring together these separate elements of the ecosystem in an integrated and quantitative way. This is done with the help of the STELLA software, which permits a dynamic specification of the individual relationships, as well as the integration of the separate relationships into a linked model. STELLA also allows the user to trace the dynamic evolution of the solution through time, rather than deriving a comparative static solution, as would be the case with the EFF model.

15.4. Scope of the Chapter

The model to be presented will be defined in more detail in Section 15.5. The overall approach is that the ecological linkage between mangrove and fishery will be elaborated, the nutrient status of the bay area and the biological productivity will be quantified, and the economic conditions relating to the level of effort, and the value of the catch, will also be elaborated. This will be done with the use of the STELLA software.

15.5. Definition of the Model

The model can be divided into three sectors: carbon stock, fish biomass, and fishery sector. The first two sectors, carbon stock and fish biomass, can be considered as the ecological part while the latter, fishery sector, is used to explain the impacts of ecological disturbance on the economic aspect.

15.5.1. Carbon Stock Sector

Three different sources of carbon discharge can be considered to be the key factor to determine the carbon stock in bay area: carbon discharge from economic activities, from mangrove area, and from shrimp farm area.

The dynamics of carbon stock is the determined by its initial value and changes during the time

$$C(t) = C(t - \mathrm{d}t) + \mathrm{CG}\,\mathrm{d}t$$

where C is the carbon stock and CG, the change in carbon stock in dt period.

$$\mathrm{CG} = \mathrm{CGE} + \mathrm{CGM} + \mathrm{CGS}$$

where CGE is the change in carbon stock from economic activities; CGM, the change in carbon stock from mangrove area; and CGS, the change in carbon stock from shrimp farm area.

$$\text{CGE} = \sum_i (\text{output}_i \times \text{emission coefficient}_i)$$

$$\text{CGM} = \text{change in mangrove area} \times \text{mangrove emission rate}$$

$$\text{CGS} = \text{change in shrimp farm area} \times \text{shrimp farm emission rate}$$

From Fig. 1 above, land use change (i.e. mangrove reforestation, shrimp farm conversion, etc.), and change in economic activities can cause a change in carbon stock in the bay area.

15.5.2. Fish Biomass Sector

Like the carbon stock sector, fish biomass depends on its initial value and changes:

$$F(t) = F(t - \mathrm{d}t) + \mathrm{FG}\,\mathrm{d}t$$

where F is the fish biomass and FG, the change in fish biomass (Fig. 2).

Change in fish biomass is equal to change in natural growth minus the catch. The change in natural growth is explained by a natural carrying capacity function.

$$\mathrm{FG} = (F \times \mathrm{rF})(\mathrm{CC} - F)/\mathrm{CC} - \text{catch}$$

where rF is the intrinsic growth rate (assumed to be 0.5) and CC, the carrying capacity.

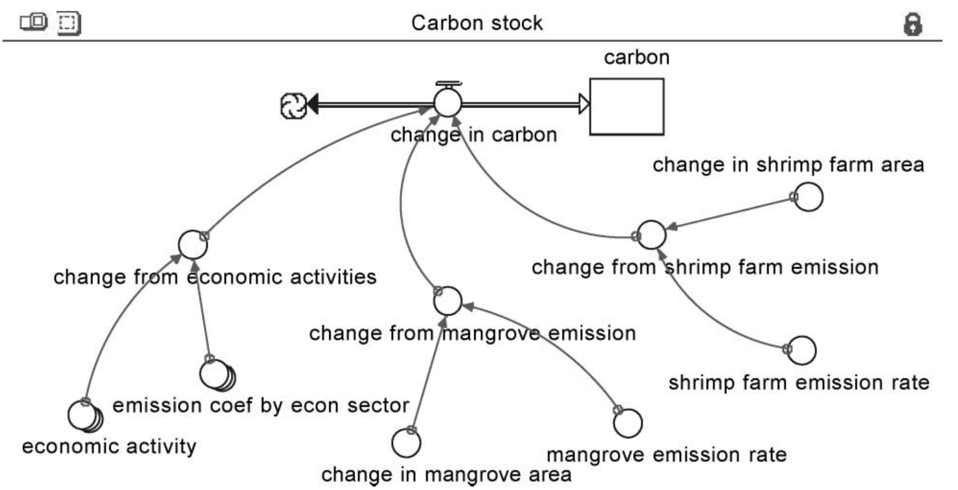

Figure 1: Carbon stock in bay area.

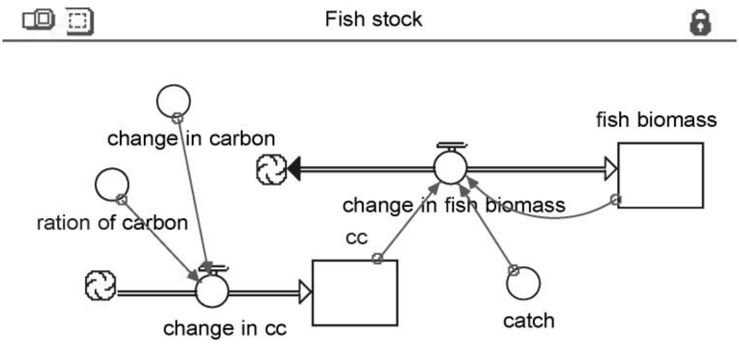

Figure 2: Fish biomass.

In this chapter, the carrying capacity for fish is not treated as a constant but is determined by the carbon stock in the area.

$$\mathrm{CC}(t) = \mathrm{CC}(t - \mathrm{d}t) + \mathrm{CCG}\,\mathrm{d}t$$

$$\mathrm{CCG} = \mathrm{CG} \times \mathrm{rC}$$

where CCG is the change in carrying capacity and RC, the carbon content in fish.

15.5.3. Fishery Sector

Fish catch is assumed to be a function of effort and the total fish biomass. The Cobb–Douglas function is used to explain the relationship in this case.

$$C = 0.35E^{0.5}F^{0.5}$$

where C is the catch and E, the effort (hours used in fishing) (Fig. 3).

Revenue from fishery is equal to the price of fish (in this chapter, assumed to be constant) times catch.

$$\mathrm{TR} = pC$$

where TR is the total revenue and p, the fish price.

Total cost of fishery in this case is equal to the cost of effort which can be estimated by wage of labor.

$$\mathrm{TC} = wE$$

where TC is the total cost and w, the wage of labor.

From the total revenue and total cost, the net revenue can be calculated from

$$\text{Net revenue} = \mathrm{TR} - \mathrm{TC}$$

$$\text{Net revenue per effort} = \text{Net revenue}/E$$

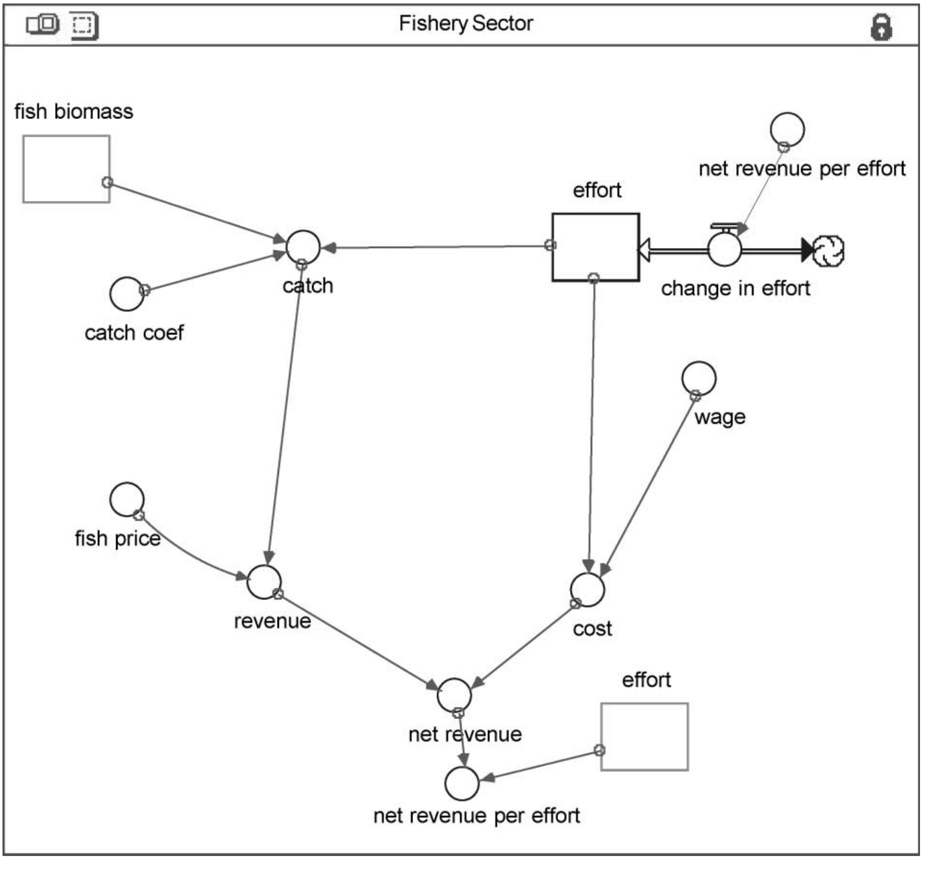

Figure 3: Fishery sector.

In the case of open access, with no barrier to entry or exit, the number of effort depends directly on the net revenue per effort. Positive net revenue per effort will encourage fishermen to increase their efforts which, instead, will gradually decrease the net revenue per effort. The increase in effort will go on until net revenue reaches zero.

The change in effort is assumed to be a linear function of net revenue per effort as follows:

$$\text{Change in effort} = a \times \text{Net revenue per effort}$$

where a is the open access coefficient which shows the relationship between new effort according to the net revenue per effort.

The coefficients, parameters, and initial values used in this chapter are shown in Appendix A.

15.6. Simulation Results

The model is simulated twice: with the baseline scenario, and the mangrove reforestation scenario, to compare the impacts of an increase in mangrove area.

15.6.1. Baseline Scenario

In the baseline scenario, the model is simulated without any change in exogenous variables, which affect carbon stock in the bay area. The outputs of key variables are shown in Figs. 4–6 (see Table B1 in Appendix B for the value of the variables).

From Fig. 4, at the beginning point, there is positive net revenue which encourage the fishermen to increase their effort in fishery. This will cause an increase in effort (as shown in Fig. 5) which will induce an increase in catch, therefore, decrease in fish biomass (Fig. 6).

15.6.2. Mangrove Reforestation Scenario

Suppose that there is an 100 km^2 increase in mangrove area, the output of key variables will be as follows (Figs. 7–9) (see Table B2 in Appendix B for the value of the variables):

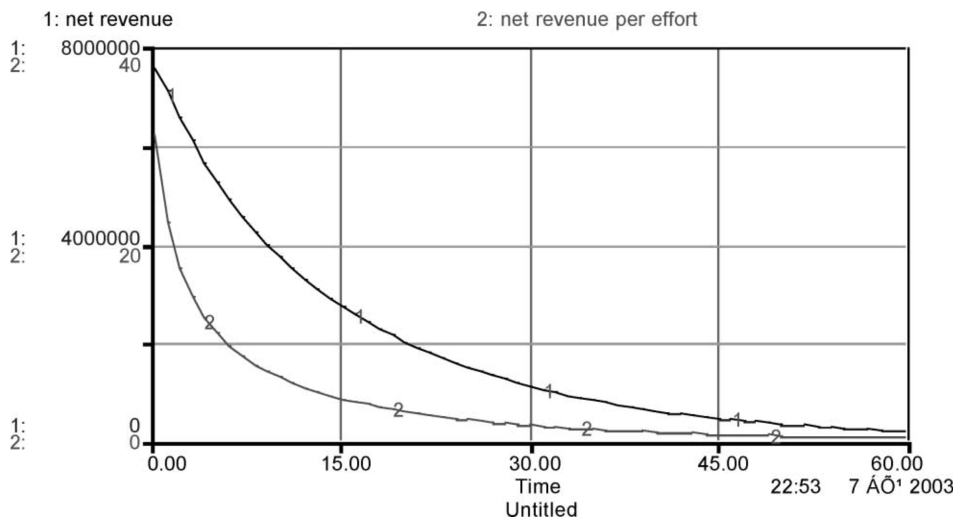

Figure 4: Net revenue and net revenue per effort (baseline case).

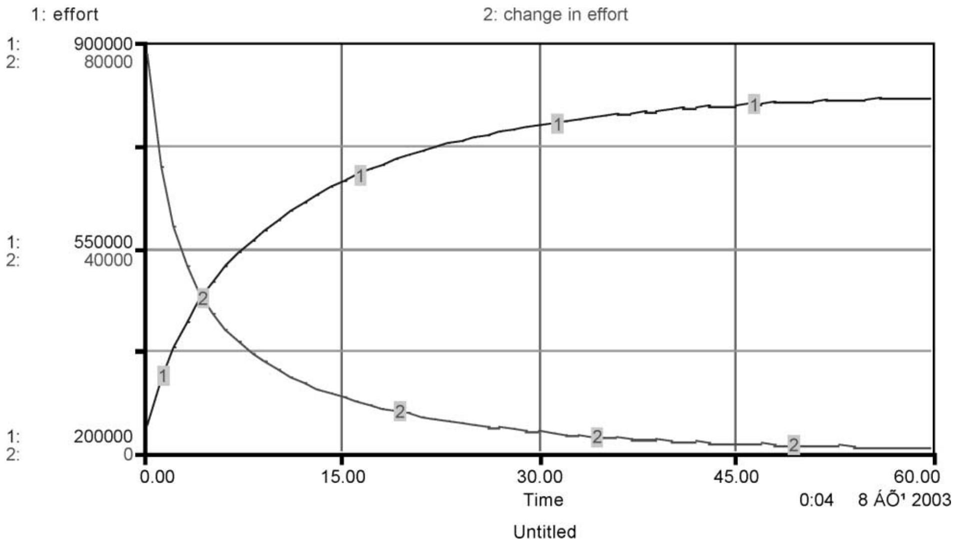

Figure 5: The effort of fishery (baseline case).

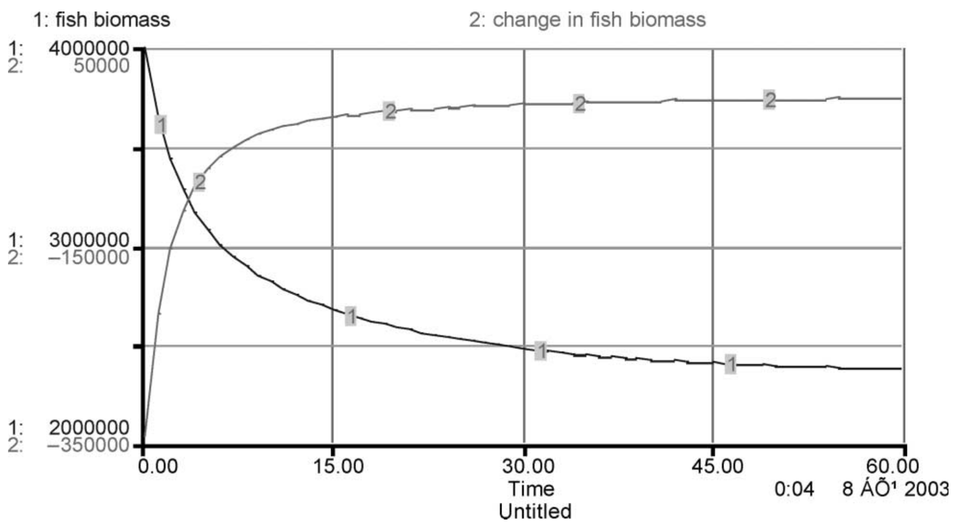

Figure 6: Fish biomass and change in fish biomass (baseline case).

Like the baseline case, positive net revenue will cause an increase in effort, catch and so decrease in fish biomass. But due to an increase in carbon stock discharged from mangrove area, fish biomass increases during that period which will lead to higher net revenue.

15.6.3. Comparison Between Baseline and Reforestation Scenario

Because of an increase in carbon stock (from mangrove area), the net revenue curve in the reforestation case will shift higher than in baseline case. The total net revenue of fishery can be presented by the area under the curve. The difference between areas under the curve is the surplus of mangrove reforestation reflecting through the net revenue of fishery sector.

From net revenue in each year, the net present value of net revenue in the baseline case is equal to 45,149,222.61 Baht, while in the reforestation case, the net present value is equal to 103,931,991.04 Baht. The surplus according to 100 km^2 reforestation will be equal to 58,782,768.44 Baht (in 60 years with 10% discount rate) (Figs. 10–12).

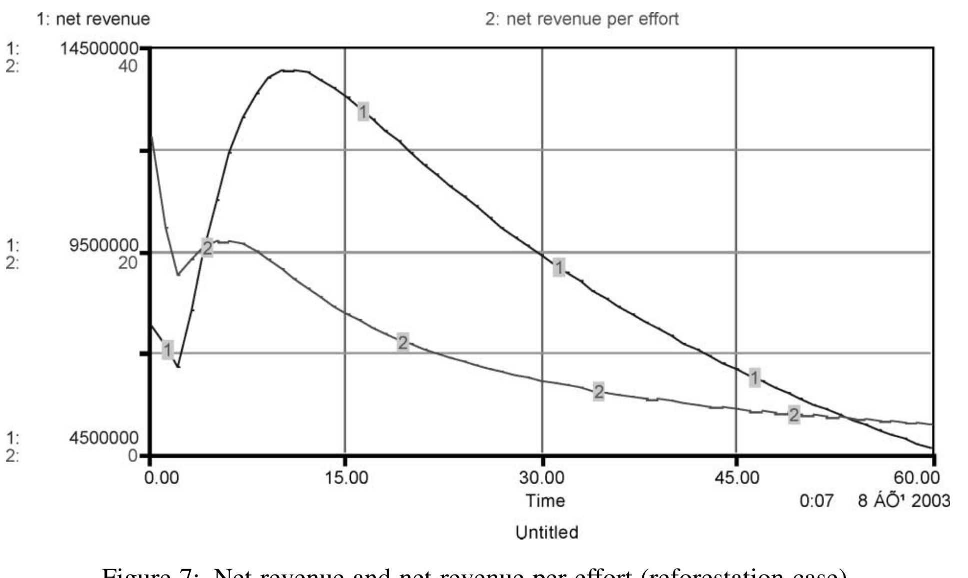

Figure 7: Net revenue and net revenue per effort (reforestation case).

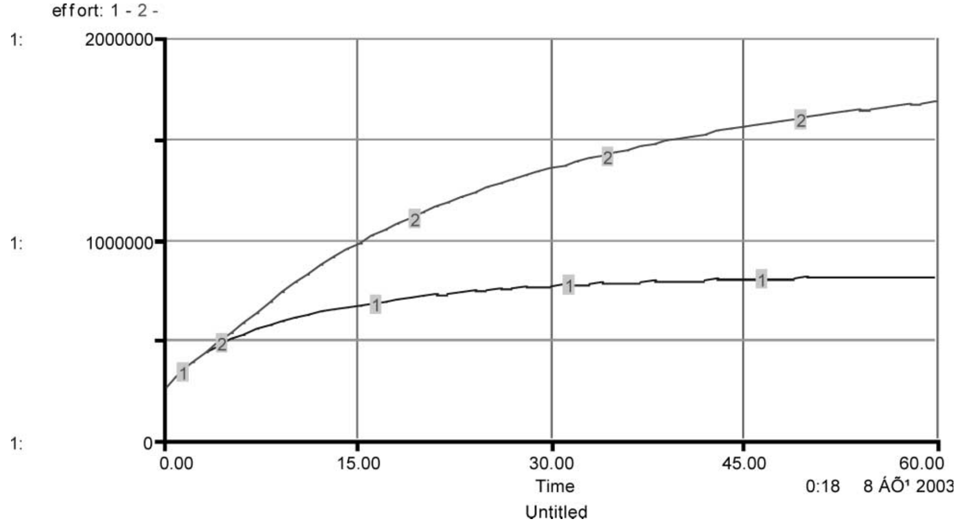

Figure 8: Effort of fishery (reforestation case).

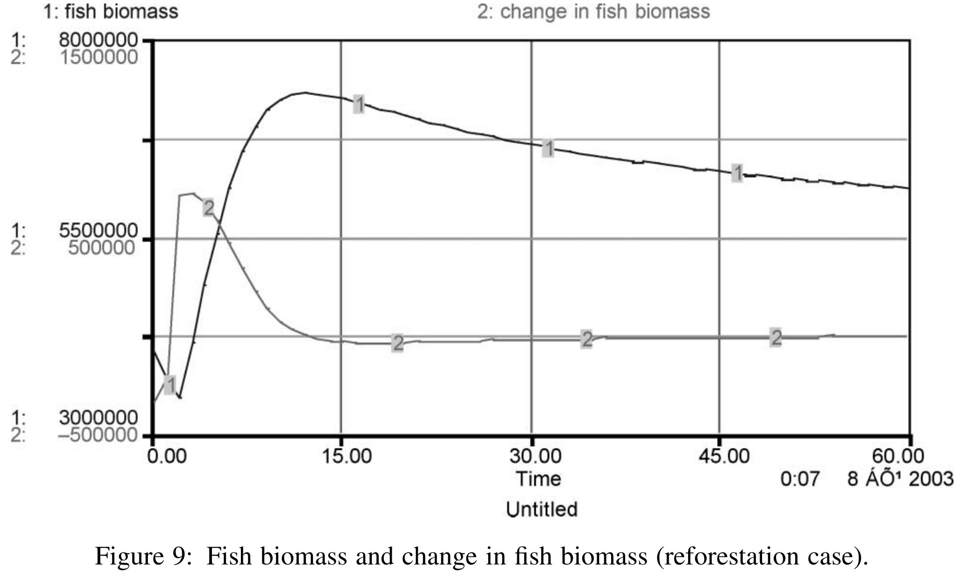

Figure 9: Fish biomass and change in fish biomass (reforestation case).

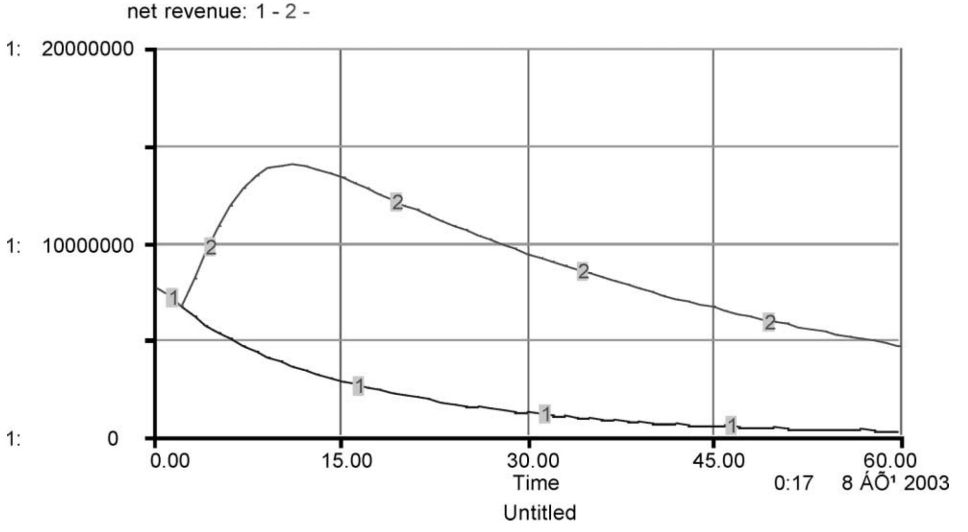

Figure 10: The comparison of net revenue between baseline and reforestation scenario.

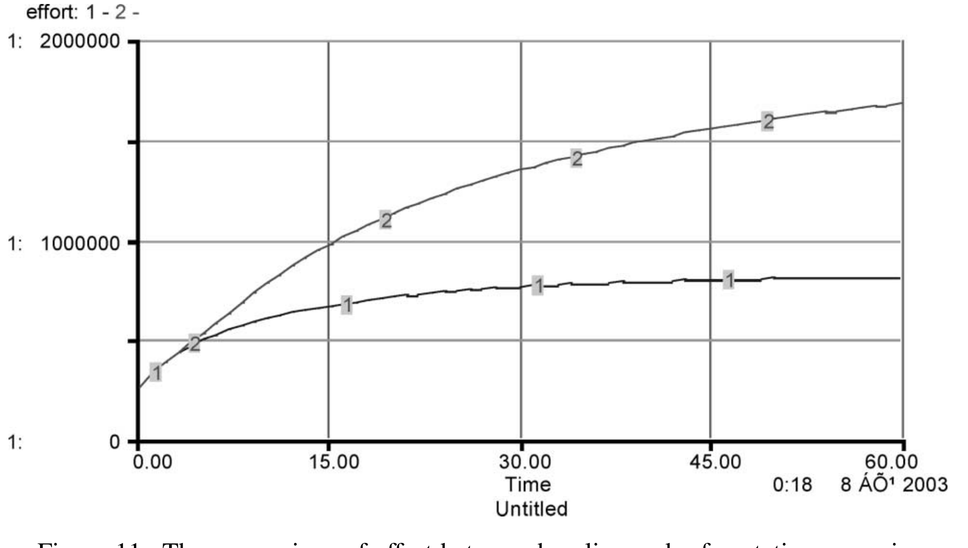

Figure 11: The comparison of effort between baseline and reforestation scenario.

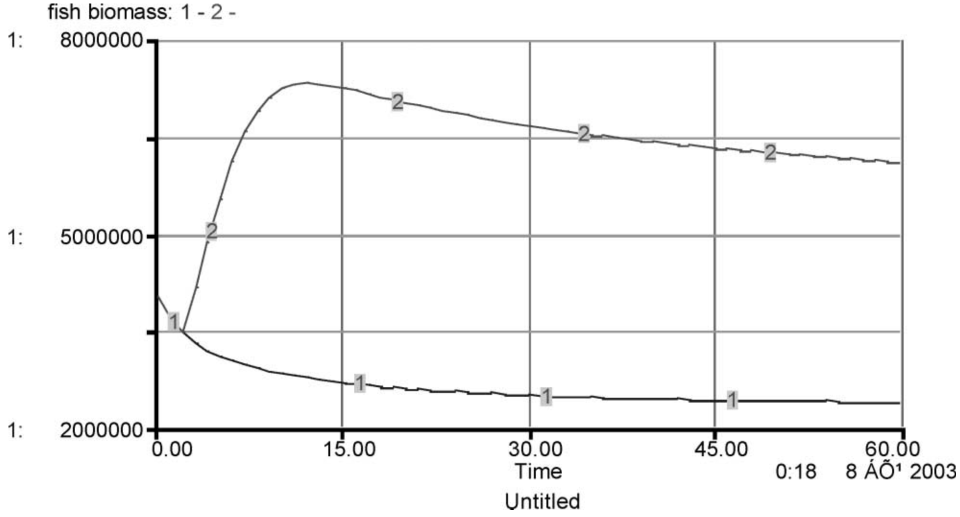

Figure 12: The comparison of fish biomass between baseline and reforestation scenario.

15.7. Discussion

The model is used to evaluate indirect use value (through fishery) of mangrove. It explores a new approach to measure the benefit by the difference between areas under net revenue curve after reforestation takes place. The simulation model incorporates the effects of interactions between the ecological factors, represented by the carbon and fish biomass relationship, and the economic factors, represented by the effort–net profit relationship. Reforestation produces a one-off gain in the yield of fish from the fishing effort, but the open access leads eventually to a position of zero net profit as before. However, there are some limitations for this study. The accuracy of the model is still questionable due to the accuracy of quantitative relationship. The value of parameters should be determined for the improvement of the model.

Appendix A

See Table A1.

Table A1: The coefficients, parameter, and initial values used in the model.

Name	Value	Source
Emission coefficient by economic sector (tonC/mB)		LOICZ (2001)
Agriculture	0.18275	
Fishery	0.09129	
Manufacturing I	0.36550	
Manufacturing II	0.18275	
Utilities	0.36550	
Construction	0.18275	
Trade	0.12036	
RestHotel	0.12036	
TransCom	0.18275	
Other services	0.12036	
Emission coefficient of mangrove area (tonC/km^2/year)	1,118	LOICZ (2001)
Emission coefficient of shrimp farm area (tonC/km^2/year)	39.44	LOICZ (2001)
Ratio of carbon in animal (%)	5.00	LOICZ (2001)
The intrinsic growth rate	0.5	Aniyar (2002)
Fish price (Baht/kg)	37.81	Sathirathai (1998)
Wage (Baht/h)	22.5	Sathirathai (1998)
Initial value of carbon (tonC)	32,837.91	LOICZ (2001)
Initial value of fish biomass (kg)	4,000,000	
Initial value of fish carrying capacity (kg)	4,000,000	
Initial value of effort (h)	243,672.2	LOICZ (2001)

Appendix B

See Tables B1 and B2.

Table B1: The simulation results of baseline scenario.

Year	Carbon (tonC)	Fish biomass (kg)	Effort (h)	Catch (kg)	Net revenue (Baht)
0	32,837.91	4,000,000.00	243,672.20	345,542.15	7,582,324.22
1	32,837.91	3,654,457.85	321,464.46	379,355.17	7,110,468.58
2	32,837.91	3,432,948.83	376,761.93	398,047.79	6,573,043.49

(*continued*)

Table B1: Continued.

Year	Carbon (tonC)	Fish biomass (kg)	Effort (h)	Catch (kg)	Net revenue (Baht)
3	32,837.91	3,278,233.25	420,377.29	410,872.99	6,076,618.73
4	32,837.91	3,163,125.23	456,515.18	420,585.07	5,630,729.84
5	32,837.91	3,073,432.62	487,350.57	428,351.84	5,230,595.06
6	32,837.91	3,001,048.59	514,182.36	434,773.58	4,869,685.93
7	32,837.91	2,941,012.73	537,859.20	440,200.77	4,542,159.06
8	32,837.91	2,890,123.84	558,971.41	444,857.66	4,243,211.16
9	32,837.91	2,846,226.12	577,949.18	448,897.90	3,968,972.93
10	32,837.91	2,807,815.90	595,117.53	452,432.43	3,716,325.89
11	32,837.91	2,773,812.65	610,729.26	455,544.67	3,482,735.66
12	32,837.91	2,743,419.73	624,985.72	458,299.33	3,266,118.87
13	32,837.91	2,716,036.29	638,050.50	460,747.89	3,064,741.51
14	32,837.91	2,691,199.90	650,058.72	462,932.12	2,877,142.38
15	32,837.91	2,668,548.12	661,123.65	464,886.47	2,702,075.42
16	32,837.91	2,647,792.07	671,341.39	466,639.71	2,538,466.20
17	32,837.91	2,628,698.04	680,794.35	468,216.13	2,385,378.76
18	32,837.91	2,611,074.26	689,553.90	469,636.42	2,241,990.19
19	32,837.91	2,594,761.37	697,682.30	470,918.35	2,107,570.83
20	32,837.91	2,579,625.39	705,234.35	472,077.27	1,981,468.91
21	32,837.91	2,565,552.42	712,258.50	473,126.54	1,863,098.26
22	32,837.91	2,552,444.69	718,797.90	474,077.80	1,751,928.64
23	32,837.91	2,540,217.50	724,891.16	474,941.26	1,647,477.84
24	32,837.91	2,528,796.87	730,572.97	475,725.92	1,549,305.30
25	32,837.91	2,518,117.68	735,874.65	476,439.74	1,457,006.85
26	32,837.91	2,508,122.20	740,824.57	477,089.74	1,370,210.39
27	32,837.91	2,498,758.94	745,448.50	477,682.19	1,288,572.25
28	32,837.91	2,489,981.69	749,769.97	478,222.65	1,211,774.10
29	32,837.91	2,481,748.78	753,810.46	478,716.10	1,139,520.41
30	32,837.91	2,474,022.45	757,589.66	479,166.98	1,071,536.17
31	32,837.91	2,466,768.31	761,125.66	479,579.27	1,007,565.02
32	32,837.91	2,459,954.95	764,435.12	479,956.57	947,367.59
33	32,837.91	2,453,553.57	767,533.38	480,302.06	890,720.00
34	32,837.91	2,447,537.65	770,434.62	480,618.66	837,412.66
35	32,837.91	2,441,882.75	773,151.96	480,908.97	787,249.09
36	32,837.91	2,436,566.23	775,697.54	481,175.34	740,044.94
37	32,837.91	2,431,567.13	778,082.64	481,419.90	695,627.10
38	32,837.91	2,426,865.96	780,317.70	481,644.57	653,832.89
39	32,837.91	2,422,444.57	782,412.47	481,851.09	614,509.33
40	32,837.91	2,418,286.05	784,375.98	482,041.05	577,512.51
41	32,837.91	2,414,374.60	786,216.65	482,215.86	542,706.97

(*continued*)

Table B1: Continued.

Year	Carbon (tonC)	Fish biomass (kg)	Effort (h)	Catch (kg)	Net revenue (Baht)
42	32,837.91	2,410,695.45	787,942.34	482,376.83	509,965.16
43	32,837.91	2,407,234.78	789,560.37	482,525.13	479,166.95
44	32,837.91	2,403,979.62	791,077.57	482,661.85	450,199.14
45	32,837.91	2,400,917.83	792,500.31	482,787.94	422,955.08
46	32,837.91	2,398,038.00	793,834.55	482,904.30	397,334.24
47	32,837.91	2,395,329.42	795,085.86	483,011.74	373,241.87
48	32,837.91	2,392,782.01	796,259.45	483,110.99	350,588.65
49	32,837.91	2,390,386.32	797,360.19	483,202.71	329,290.37
50	32,837.91	2,388,133.42	798,392.63	483,287.54	309,267.65
51	32,837.91	2,386,014.93	799,361.04	483,366.01	290,445.65
52	32,837.91	2,384,022.98	800,269.40	483,438.65	272,753.81
53	32,837.91	2,382,150.12	801,121.47	483,505.92	256,125.62
54	32,837.91	2,380,389.36	801,920.75	483,568.24	240,498.38
55	32,837.91	2,378,734.12	802,670.50	483,626.01	225,813.00
56	32,837.91	2,377,178.17	803,373.82	483,679.58	212,013.78
57	32,837.91	2,375,715.67	804,033.58	483,729.27	199,048.23
58	32,837.91	2,374,341.12	804,652.49	483,775.40	186,866.89
59	32,837.91	2,373,049.31	805,233.07	483,818.23	175,423.15
Final	32,837.91	2,371,835.36	805,777.70	483,858.01	164,673.10

Table B2: The simulation results of reforestation scenario.

Year	Carbon (tonC)	Fish biomass (kg)	Effort (h)	Catch (kg)	Net revenue (Baht)
0	32,837.91	4,000,000.00	243,672.20	345,542.15	7,582,324.22
1	32,837.91	3,654,457.85	321,464.46	379,355.17	7,110,468.58
2	144,637.91	3,432,948.83	376,761.93	398,047.79	6,573,043.49
3	144,637.91	4,136,926.16	420,377.29	461,558.29	7,993,029.76
4	144,637.91	4,851,539.08	467,912.15	527,339.10	9,410,667.95
5	144,637.91	5,522,783.32	518,192.25	592,096.51	10,727,843.52
6	144,637.91	6,101,821.13	569,948.35	652,702.88	11,854,858.14
7	144,637.91	6,558,828.55	621,948.05	706,900.42	12,734,073.70
8	144,637.91	6,888,473.17	673,134.29	753,668.54	13,350,685.97
9	144,637.91	7,105,054.63	722,718.33	793,115.23	13,726,524.45
10	144,637.91	7,232,464.56	770,200.61	826,062.92	13,903,925.47
11	144,637.91	7,295,389.73	815,331.46	853,609.80	13,930,028.72

(*continued*)

Table B2: Continued.

Year	Carbon (tonC)	Fish biomass (kg)	Effort (h)	Catch (kg)	Net revenue (Baht)
12	144,637.91	7,314,568.05	858,044.24	876,833.69	13,847,086.31
13	144,637.91	7,305,502.96	898,389.15	896,654.96	13,688,768.08
14	144,637.91	7,278,993.99	936,481.68	913,804.66	13,480,116.31
15	144,637.91	7,242,238.28	972,467.75	928,842.44	13,239,008.46
16	144,637.91	7,199,894.78	1,006,502.32	942,190.04	12,977,903.25
17	144,637.91	7,154,915.69	1,038,737.47	954,164.39	12,705,362.44
18	144,637.91	7,109,136.23	1,069,316.33	965,005.00	12,427,221.50
19	144,637.91	7,063,672.34	1,098,370.46	974,894.77	12,147,435.80
20	144,637.91	7,019,181.68	1,126,019.23	983,975.28	11,868,672.84
21	144,637.91	6,976,032.30	1,152,370.18	992,357.81	11,592,719.69
22	144,637.91	6,934,410.84	1,177,519.91	1,000,131.16	11,320,761.15
23	144,637.91	6,894,391.58	1,201,555.09	1,007,367.33	11,053,569.13
24	144,637.91	6,855,980.45	1,224,553.56	1,014,125.54	10,791,631.77
25	144,637.91	6,819,142.95	1,246,585.33	1,020,455.21	10,535,241.61
26	144,637.91	6,783,821.75	1,267,713.53	1,026,398.05	10,284,556.13
27	144,637.91	6,749,947.84	1,287,995.23	1,031,989.74	10,039,639.44
28	144,637.91	6,717,447.35	1,307,482.18	1,037,261.05	9,800,491.18
29	144,637.91	6,686,245.80	1,326,221.42	1,042,238.79	9,567,066.66
30	144,637.91	6,656,270.53	1,344,255.87	1,046,946.52	9,339,290.85
31	144,637.91	6,627,452.12	1,361,624.75	1,051,405.06	9,117,068.35
32	144,637.91	6,599,725.07	1,378,364.07	1,055,632.95	8,900,290.29
33	144,637.91	6,573,028.11	1,394,506.92	1,059,646.79	8,688,839.32
34	144,637.91	6,547,304.21	1,410,083.83	1,063,461.50	8,482,593.05
35	144,637.91	6,522,500.43	1,425,122.99	1,067,090.55	8,281,426.57
36	144,637.91	6,498,567.71	1,439,650.56	1,070,546.20	8,085,214.21
37	144,637.91	6,475,460.62	1,453,690.79	1,073,839.56	7,893,830.76
38	144,637.91	6,453,137.07	1,467,266.29	1,076,980.80	7,707,152.37
39	144,637.91	6,431,558.07	1,480,398.11	1,079,979.23	7,525,057.15
40	144,637.91	6,410,687.43	1,493,105.94	1,082,843.41	7,347,425.60
41	144,637.91	6,390,491.52	1,505,408.19	1,085,581.20	7,174,140.83
42	144,637.91	6,370,939.09	1,517,322.14	1,088,199.87	7,005,088.83
43	144,637.91	6,352,001.01	1,528,864.00	1,090,706.12	6,840,158.48
44	144,637.91	6,333,650.09	1,540,049.03	1,093,106.19	6,679,241.64
45	144,637.91	6,315,860.95	1,550,891.61	1,095,405.83	6,522,233.14
46	144,637.91	6,298,609.81	1,561,405.30	1,097,610.42	6,369,030.79
47	144,637.91	6,281,874.39	1,571,602.89	1,099,724.95	6,219,535.30
48	144,637.91	6,265,633.78	1,581,496.51	1,101,754.08	6,073,650.25
49	144,637.91	6,249,868.30	1,591,097.62	1,103,702.15	5,931,282.00

(*continued*)

Table B2: Continued.

Year	Carbon (tonC)	Fish biomass (kg)	Effort (h)	Catch (kg)	Net revenue (Baht)
50	144,637.91	6,234,559.43	1,600,417.10	1,105,573.25	5,792,339.65
51	144,637.91	6,219,689.71	1,609,465.27	1,107,371.16	5,656,734.95
52	144,637.91	6,205,242.65	1,618,251.94	1,109,099.47	5,524,382.21
53	144,637.91	6,191,202.65	1,626,786.43	1,110,761.52	5,395,198.24
54	144,637.91	6,177,554.96	1,635,077.62	1,112,360.45	5,269,102.26
55	144,637.91	6,164,285.58	1,643,133.97	1,113,899.24	5,146,015.84
56	144,637.91	6,151,381.23	1,650,963.55	1,115,380.66	5,025,862.79
57	144,637.91	6,138,829.30	1,658,574.05	1,116,807.33	4,908,569.11
58	144,637.91	6,126,617.79	1,665,972.82	1,118,181.74	4,794,062.92
59	144,637.91	6,114,735.28	1,673,166.91	1,119,506.21	4,682,274.37
Final	144,637.91	6,103,170.87	1,680,163.04	1,120,782.97	4,573,135.59

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Session IV

Constructed Wetlands

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Chapter 16

Constructed Wetlands for Wastewater Treatment: Principles and Practices

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Abstract. Constructed wetlands (CW), either free water surface or subsurface flow type, are natural treatment systems which employ activities of microbes, media or plants, in waste stabilization without the aid of mechanical or energy-intensive equipment. CW can significantly reduce biochemical oxygen demand (BOD_5), suspended solids (SS), and nitrogen, as well as metals, trace organics, and pathogens. The basic treatment mechanisms include sedimentation, chemical precipitation and adsorption, and microbial interactions with BOD_5, SS and nitrogen, as well as some uptake by the vegetation. The process stability under varying environmental conditions, lower construction and operating costs, and the possibility to create a wildlife habitat, in the case of FWS systems are advantages over the conventional wastewater treatment systems. The factors affecting wastewater treatment efficiency of CW are the area of CW, depth, porosity of the material, type of plants which transfer oxygen to the bulk through roots, hydraulic budget, site selection, flow pattern. Two operational considerations associated with wetlands for wastewater treatment are mosquito control and plant harvesting.

16.1. Introduction

Wetlands are areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to maintain saturated conditions (U.S. EPA, 1988). They are comparatively shallow (typically less than 0.6 m) bodies of slow-moving water in which dense stands of water-tolerant plants such as cattails, bulrushes, or reeds are grown. In man-made systems, these bodies are artificially created and are typically long, narrow trenches or channels.

Constructing a wetland where one did not exist before avoids many of the environmental concerns and user conflicts associated with natural wetlands and allows design of the wetland for optimum wastewater treatment. Unlike natural wetlands, which are confined by availability and proximity of the wastewater

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source, constructed wetlands (CW) can be built almost anywhere, including lands with limited uses. Typically, a CW should perform better than a natural wetland of equal area since the bottom is usually graded and the hydraulic regime in the system is controlled. Process reliability is also improved because the vegetation and the other system components can be managed as required (Reed et al., 1988).

16.2. Types and Functions of Constructed Wetlands

CW are classified into two types: free water surface (FWS) systems with shallow water depths and subsurface flow (SF) or vegetated submerged bed (VSB) systems with water flowing laterally through the sand or gravel.

16.2.1. Free Water Surface Systems

These systems typically consist of basins or channels, with a natural or constructed subsurface barrier of clay or impervious geotechnical material to prevent seepage, soil or another suitable medium to support the emergent vegetation, and water at a relatively shallow depth flowing over the soil surface (Fig. 1). The shallow water depth, low flow velocity and presence of the plant stalks and litter regulate water flow and, especially in long, narrow channels, ensure plug-flow conditions to minimize short-circuiting (U.S. EPA, 1988).

16.2.2. Subsurface Flow (SF) Systems

An SF system typically consists of a trench or a bed underlain by impermeable material to prevent seepage and containing a medium that supports the growth of

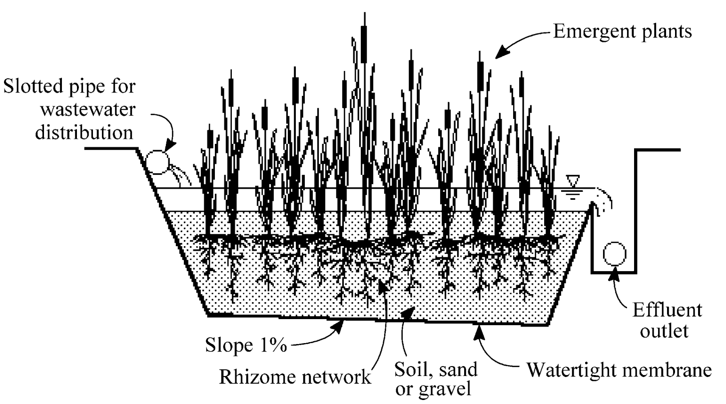

Figure 1: Free water surface (FWS) system.

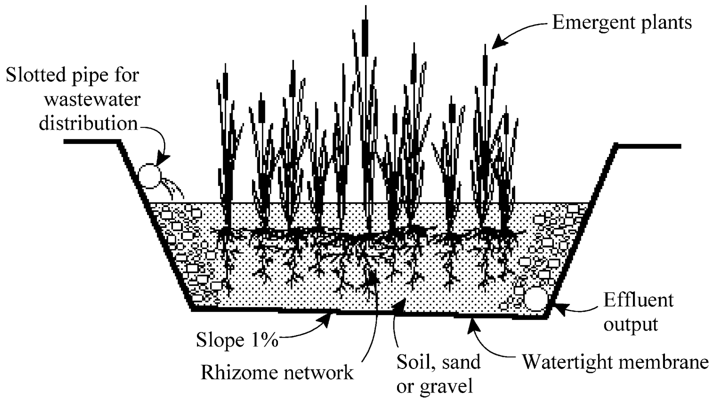

Figure 2: Subsurface flow (SF) system.

emergent vegetation (Fig. 2). The media used have included rock or crushed stone (10–15 cm diameter), gravel, and different soils, either alone or in various combinations (Reed et al., 1988). The wastewater flows laterally through the medium and is purified during the contact with the surfaces of the medium and the root zone of the vegetation. This subsurface zone is continuously saturated and therefore is generally anaerobic. However, the plants can convey an excess of oxygen to the root system, so there are aerobic microsites adjacent to the roots and rhizomes.

16.2.3. Advantages and Disadvantages

CW offer several potential advantages as a wastewater treatment process. These potential advantages include site location flexibility, less rigorous pre-application treatment, no alteration of natural wetlands, simple operation and maintenance, process stability under varying environmental conditions, lower construction and operating costs, and in the case of FWS systems, the possibility to create a wildlife habitat. The potential problems with FWS CW include mosquitoes. Start-up problems in establishing the desired aquatic plant species can be a problem with FWS and SF wetlands alike (U.S. EPA, 1988; Bastian et al., 1989).

16.3. Types and Functions of Vegetation

The major benefit of plants in CW is the transferring of oxygen to the root zone. The plant roots in the system penetrate the soil or support medium, and transport oxygen deeper than it would naturally travel by diffusion alone.

Perhaps most important in the FWS wetlands are the submerged portions of the leaves, stalks, and litter, which serve as the substrate for attached microbial

Table 1: Emergent aquatic plants for wastewater treatment.

Common name	Scientific name	Temperature (°C)		Maximum salinity tolerance (ppt[a])	Root penetration (cm)	Effective pH range
		Desirable	Seed germination			
Cattail	*Typha* spp.	10–30	12–24	30	30	4–10
Common reed	*Phragmites communis*	12–23	10–30	45	60	2–8
Rush	*Juncus* spp.	16–26	–	20	–	5–7.5
Bulrush	*Scirpus* spp.	16–27	–	20	76	4–9
Sedge	*Carex* spp.	14–32	–	–	–	5–7.5

Modified from U.S. EPA (1988).
[a] ppt denotes parts per thousand.

growth. It is the responses of this attached biota that is believed responsible for much of the treatment that occurs.

The emergent plants most frequently found in wastewater wetlands include cattails, reeds, rushes, bulrushes and sedges. It is estimated that these plants transfer 5–45 g O_2/day/m^2 depending on plant density and oxygen stress levels in the soil (Reed et al., 1988). Table 1 lists some of the major environmental requirements of each as well as their maximum depths of root penetration.

16.4. Wastewater Treatment Mechanisms

Wetland systems can significantly reduce biochemical oxygen demand (BOD_5), suspended solids (SS), and nitrogen, as well as metals, trace organics, and pathogens. The basic treatment mechanisms include sedimentation, chemical precipitation and adsorption, and microbial interactions with BOD_5, SS and nitrogen, as well as some uptake by the vegetation (U.S. EPA, 1988).

16.4.1. BOD Removal

The removal of settleable organics is very rapid in all wetland systems and is due to the quiescent conditions in the FWS systems, and to deposition and filtration in the SF systems.

In FWS wetlands, removal of the soluble BOD is mainly due to the attached microbial growth. The major source of oxygen for these reactions is reaeration at the water surface, since algae are typically not present. Any excess oxygen transmitted by the plant to the root zone is likely to be consumed in the soil profile and not to contribute significantly to oxygen levels in the water. Wind-induced water turbulence and mixing will also be reduced or eliminated if a dense stand of vegetation is present.

The major oxygen source for the subsurface components (soil, gravel, rock and other media, in trenches or beds) are the gases transmitted by the vegetation to the root zone. In most cases the system is designed to maintain flow below the surface of the bed, so there can be very little direct atmospheric reaeration. The selection of plant species can therefore be an important factor.

Table 2 reports the removal efficiencies of BOD_5 and SS from CW in Canada, U.S.A. and Australia.

16.4.2. Suspended Solids Removal

SS removal is very effective in both types of CW (see Table 2). Most of the removal occurs within the first few meters beyond the inlet, owing to the quiescent conditions and the shallow depth of liquid in the system. Controlled dispersion of

Table 2: Summary of BOD_5 and SS removal from constructed wetlands.

Project	**Flow (m^3/day)**	**Wetland type**	**BOD_5 (mg/l)**		**SS (mg/l)**		**% Reduction**		**Hydraulic surface loading rate (m^3/ha/day)**
			Inf.	**Eff.**	**Inf.**	**Eff.**	**BOD_5**	**SS**	
Listowel, Ontario	17	FWS[a]	56	10	111	8	82	93	–
Santee, CA	–	SF[b]	118	30	57	5.5	75	90	–
Sydney, Australia	240	SF	33	4.6	57	4.5	86	92	–
Arcata, CA	11,350	FWS	36	13	43	31	64	28	907
Emmitsburg, MD	132	SF	62	18	30	8.3	71	73	1,543
Gustine, CA	3,785	FWS	150	24	140	19	84	86	412

U.S. EPA (1988).
[a] Free water surface system.
[b] Subsurface flow system.

the influent flow with proper diffuser pipe design can help to ensure low velocities for solids removal, and even loading of the wetland so that anoxic conditions are prevented at the upstream end of the channels.

16.4.3. Nitrogen Removal

Nitrogen is mainly removed by nitrification/denitrification. Other removal mechanisms include plant uptake and volatilization as ammonia. In CW, nitrogen removal ranges from 25 to 85% (U.S. EPA, 1988).

16.4.4. Phosphorus Removal

Phosphorus removal in wetlands is not very effective because of the limited contact opportunities between the wastewater and the soil. The principal mechanisms for phosphorus removal are plant uptake or retention in the soil (U.S. EPA, 1988).

If phosphorus removal is required, clay with iron and aluminum content should be considered. However, soils with a high phosphorus removal capacity are finer textured, and sand may be added to improve hydraulic conductivity. Also, iron or aluminum added to the substrate or fed into the wastewater can improve phosphorus removal (Steiner & Freeman, 1989).

16.4.5. Heavy Metals Removal

The predominant removal mechanisms in CW are precipitation and adsorption. Precipitation is enhanced by wetland metabolism which increases the pH of inflowing acidic waters to near neutrality. Removal of Cu, Zn and Cd at the rates of 99, 97 and 99%, respectively, for a residence time of 5.5 days in the Santee, CA, wetlands have been reported (U.S. EPA, 1988). However, metals removal will likely be finite due to exhaustion of the soil cation exchange capacity (CEC).

16.4.6. Trace Organics Removal

Municipal and industrial wastewaters contain variable concentrations of synthetic organic compounds. Adsorption of trace organics by the organic matter and clay particles present in the treatment system is thought to be the primary physicochemical mechanism for removal of refractory compounds in wetlands.

Other mechanisms can be biological degradation of easily degraded organic compounds, sedimentation and volatilization (U.S. EPA, 1988).

16.4.7. Pathogen Removal

The pathogens of concern in CW are parasites, bacteria and viruses. Pathogenic bacteria and viruses are removed by such mechanisms predation, sedimentation, absorption, and die-off from unfavorable environmental conditions, including UV in sunlight and temperatures unfavorable for cell reproduction (U.S. EPA, 1988). Table 3 reports performance data on pathogen removal for both FWS and SF wetlands in the US and Canada.

16.5. Design Equations

All CW systems can be considered to be attached growth biological reactors, and their performance can be described with first order plug-flow kinetics. Design equations given below for both FWS and SF systems should give a reasonable, and hopefully conservative, estimate of design requirements. However, a pilot test is strongly recommended for large-scale projects.

16.5.1. FWS Wetlands

BOD_5 removal in plug-flow reactors such as a wetland can be described by first-order model as follows (Reed et al., 1988):

$$\frac{C_e}{C_0} = \exp(-K_T t) \tag{1}$$

where C_e is the effluent BOD_5, mg/l; C_0, influent BOD_5, mg/l; K_T, temperature-dependent first-order reaction rate constant, $= k_{20}(1.06)^{T-20}\text{day}^{-1}$; t, hydraulic residence time, day (in calculating the t value, the wetland bed porosity and water loss by evapotranspiration (ET) must be taken into account); T, water temperature, °C; $k_{20} = 0.678\ \text{day}^{-1}$.

For an unrestricted flow system, hydraulic residence time can be expressed as:

$$t = \frac{LWd}{Q} \tag{2}$$

where L is the length of system, m; W, width of system, m; d, depth of system including the bed depth and water depth, m; Q, average flow rate $=$ $(Q_{\text{inf}} + Q_{\text{eff}})/2$.

Table 3: Pathogen removal in constructed wetland systems.

Location (vegetation)	Winter season			Summer season		
	Influent	**Effluent**[a]	**% Reduction**	**Influent**	**Effluent**[a]	**% Reduction**
Santee, CA (bulrush)[b]						
Total coli (no./100 ml)	5×10^7	1×10^5	99.80	6.5×10^7	3×10^5	99.54
Bacteriophage (PFU/ml)	1,900	15	99.21	2,300	26	98.87
Iselin, PA (cattails and grasses)[c]						
Fecal coli (no./100 ml)	1.7×10^6	6,200	99.64	1.0×10^6	723	99.93
Arcata, CA (bulrush)[d]						
Fecal coli (no./100 ml)	4,300	900	79.07	1,800	80	95.56
Listowel, Ont. (cattails)[d]						
Fecal coli (no./100 ml)	556,000	1,400	99.75	198,000	400	99.80

Modified from Reed et al. (1988).
[a] Undisinfected.
[b] Gravel bed, subsurface flow.
[c] Sand bed, subsurface flow.
[d] Free water surface.

In an FWS CW, because a portion of the available volume will be occupied by the bed media and the vegetation, the actual retention will be a function of the porosity (η), which can be defined as the remaining cross-sectional area available for flow:

$$\eta = \frac{V_V}{V} \tag{3}$$

where V_V and V are volume of voids and total volume, respectively.

The value of η for FWS CW is generally taken as 0.75. So the actual retention time is:

$$t = \frac{LWd\eta}{Q} \tag{4}$$

Reed et al. (1988) developed another general model for FWS CW design by combining the relationships in Eqs. (2) and (3) with Eq. (1):

$$\frac{C_e}{C_0} = A\ \exp\left[-\frac{0.7K_T(A_V)^{1.75}LWd\eta}{Q}\right] \tag{5}$$

where C_e is the effluent BOD_5, mg/l; C_0, influent BOD_5, mg/L; A, fraction of BOD_5 not removed as settleable solids near headworks of the system (as a decimal fraction); K_T, temperature-dependent rate constant, day^{-1}; A_V, specific surface area for microbial activity, m^2/m^3; L, length of system (parallel to flow path), m; W, width of system, m; d, design depth of system, m; η, porosity of system (as a decimal fraction); Q, average flow rate = $(Q_{inf} + Q_{eff})/2$.

The rate constant K_T (in day^{-1}) at water temperature T (in °C) and is defined by:

$$K_T = K_{20}(1.1)^{(T-20)} \tag{6}$$

where, K_{20} is the rate constant at 20°C.

Other coefficients in Eq. (5) have been estimated (Reed et al., 1988):

$$A = 0.52;\ K_{20} = 0.0057\ \text{day}^{-1};\ A_V = 15.7\ \text{m}^2/\text{m}^3;\ \eta = 0.75$$

When the bed slope or hydraulic gradient (S) is greater than 1%, it is necessary to adjust the design model accordingly:

$$\frac{C_e}{C_0} = A\ \exp\left[-\frac{0.7K_T(A_V)^{1.75}LWd\eta}{4.63S^{1/3}Q}\right] \tag{7}$$

Table 4 lists typical values used to test the equation against actual values at Listowel, Ontario, Canada. Table 5 summarizes design criteria for FWS wetlands. It should be noted in this table that an aspect ratio (L/W) of at least 10:1 is needed for FWS wetland systems to ensure plug-flow conditions and optimum treatment.

Table 4: Predicted vs. actual C_e/C_0 values for constructed wetlands at Listowel, Ontario.

Distance along channel (m)	**Fall**		**Winter**		**Spring**		**Summer**	
	Predicted	**Actual**	**Predicted**	**Actual**	**Predicted**	**Actual**	**Predicted**	**Actual**
0	0.52	0.52	0.52	0.52	0.52	0.52	0.52	0.52
67	0.38	0.40	0.40	0.40	0.47	0.30	0.38	0.36
134	0.27	0.23	0.31	0.20	0.42	0.28	0.27	0.41
200	0.20	0.19	0.24	0.19	0.38	0.22	0.20	0.30
267	0.14	0.18	0.18	0.17	0.34	0.23	0.14	0.27
334 (final effluent)	0.10	0.15	0.14	0.17	0.30	0.26	0.10	0.17

Reed et al. (1988).

Table 5: Summary of design criteria for FWS constructed wetlands.

Organic loading (kg BOD_5/ha/day)	< 112
Actual retention time as determined by Eq. (5) or (7) (days)	3–15
Specific surface (A_V in Eq. (5) or (7)) for attached microbial growth (m^2/m^3)	15.7
Porosity (η value in Eq. (5) or (7)) of wetland flow path	0.75
Aspect ratio (L/W)	$\geq$10:1
Water depth (cm)	
Warm months	<10
Cool months	<45

Reed et al. (1988).

Substituting these factors in Eq. (5) or (7) and rearranging terms to solve for retention time or for the required surface area produced the following equations for FWS wetlands.

Hydraulic residence time is given by:

$$t = \frac{(\ln C_0 - \ln C_e) - 0.6539}{65K_T} \tag{8}$$

If the bed slope or hydraulic gradient, is greater than 1%, then

$$t = \frac{(\ln C_0 - \ln C_e) - 0.6539}{301K_T S^{1/3}} \tag{9}$$

where, S is the bed slope in decimal fraction (e.g. for a 2% bed slope, S is equal to 0.02).

The design surface area of the wetland is given by

$$A = \frac{Q(\ln C_0 - \ln C_e - 0.6539)}{65K_T d} \tag{10}$$

If the bed slope or hydraulic gradient is greater than 1%

$$A = \frac{Q(\ln C_0 - \ln C_e - 0.6539)}{301K_T S^{1/3}} \tag{11}$$

Eqs. (8)–(11) are only valid for FWS wetlands meeting the conditions defined in the criteria summarized in Table 5. Design of wetlands with large unvegetated areas can use the general form presented in Eq. (5) or (7).

Example 1. Design an FWS wetland to produce advanced secondary effluent in a warm climate with a mean annual temperature of 25°C. The design flow is 760 m^3/day, influent wastewater is from a facultative lagoon with a BOD_5 concentration of 130 mg/l, and required effluent BOD_5 is 10 mg/l.

Solution

1. Assume the slope of the wetland bed will be 1% to allow drainage when required. Use Eq. (8) to estimate required retention time. At 25°C,

$$K_T = K_{20}(1.1)^{(T-20)} = 0.0057(1.1)^{(25-20)} = 0.0092\ \text{day}^{-1}$$

$$t = \frac{(\ln C_0 - \ln C_e) - 0.6539}{65K_T}$$

$$t = \frac{(\ln 130 - \ln 10) - 0.6539}{(65)(0.0092)} = 3.2\ \text{days}$$

2. For the warm climate site, use a 10-cm water depth on a year-round basis. If cattail plants are to be grown, the bed depth should be 30 cm, the total bed depth or d is 40 cm. Use Eq. (10) to estimate the surface area required.

$$A = \frac{Q(\ln C_0 - \ln C_e - 0.6539)}{65K_T d} = \frac{(760\ \text{m}^3/\text{day})(\ln 130 - \ln 10 - 0.6539)}{(65)(0.0092)(0.40\ \text{m})(10,000\ \text{m}^2/\text{ha})}$$

$$= 0.60\ \text{ha}$$

3. Use an aspect ratio (L/W) of 10:1 and determine the dimensions for the wetland channels, assuming a square plot is available.

$$A = LW = (10W)W = 600\ \text{m}^2$$

   Thus $W = 7.75$ m, $L = 77.5$ m. Divide the square plot into 10 parallel channels, each 77.5 m long.
4. The design procedure assumes, and experience at operational systems confirms that SS and nitrogen concentrations in the effluent will also satisfy advanced secondary treatment requirements. Some of the remaining nitrogen will be in the ammonia form. If stringent ammonia limits prevail, recycle of wetland effluent should be incorporated in the system design. The amount of recycle required will depend on the ET losses for the area and should be sufficient to maintain the 3.2-day retention time.

16.5.2. SF Wetlands

The major oxygen source for the subsurface components (soil, gravel, rock, and other media in trenches or beds) is the oxygen transmitted by the vegetation to the root zone. In most cases, the SF system is designed to maintain flow below the surface of the bed, so there can be very little direct atmospheric reaeration.

The selection of plant species is therefore an important factor. The root penetration of the various plants (see Table 1) makes it possible to remove BOD_5 in the expanded aerobic zone.

Root penetration is important for the oxygen transfer and treatment of organics and nitrogen at the full bed depth. The maximum treatment potential, and possibly the maximum design hydraulic loading, may not be realized until the roots and rhizomes have penetrated to their full potential depth. Reed et al. (1988) recommended that the depth of root penetration, as listed in Table 1, be used as the design depth for SF systems.

BOD_5 removal in SF systems can be described with first-order kinetics, as described in Eq. (1) for FWS systems. Eq. (1) can be rearranged and used to estimate the required surface area for an SF system. Both forms of the equation are shown below for convenience.

$$\frac{C_e}{C_0} = \exp(-K_T t) \tag{1}$$

$$A_s = \frac{Q(\ln C_0 - \ln C_e)}{K_T d \eta} \tag{12}$$

where C_e is the effluent BOD_5, mg/l; C_0, influent BOD_5, mg/l; K_T, temperature-dependent first-order reaction rate constant, $= k_{20}(1.06)^{T-20}$ day^{-1}; t, hydraulic residence time, day; Q, average flow rate through the system, m^3/day; d, depth of submergence, m; η, porosity of the bed, as a fraction; A_s, surface area of the system, m^2.

The saturated cross-sectional area for flow through an SF system can be calculated according to Darcy's law:

$$A_c = \frac{Q}{k_s S} \tag{13}$$

where A_c is the $d{\cdot}W$, cross-sectional area of wetland bed, perpendicular to the flow direction, m^2; d, bed depth, m; W, bed width, m; k_s, hydraulic conductivity of the medium, m^3/m^2 day; S, slope of the bed, or hydraulic gradient (as a decimal fraction).

Bed cross-sectional area and bed width are independent of temperature (climate) and organic loading since they are controlled by the hydraulic characteristics of the media.

The value of K_T can be calculated from suggested k_{20} values for SF CW systems (Table 6). Table 6 lists expected porosities (η), hydraulic conductivity and k_{20}.

To avoid disruption of the medium-rhizome structure and to ensure sufficient contact time for treatment, the unit flow velocity (Q/A_c), which is equal to $k_s S$ according to Eq. (13) through a cross-section of the medium should not exceed 8.6 m/day (Reed et al., 1988).

Table 6: Media characteristics for subsurface flow systems.

Media type	**Maximum 10% grain size (mm)**	**Porosity (η)**	**Hydraulic conductivity (k_s) (m^3/m^2/day)**	k_{20}
Medium sand	1	0.42	420	1.84
Coarse sand	2	0.39	480	1.35
Gravelly sand	8	0.35	500	0.86

U.S. EPA (1988).

Example 2. Design an SF wetland to produce advanced secondary effluent in a warm climate with a mean annual temperature of 25°C. The design flow is 760 m^3/day, influent wastewater is from a facultative lagoon with a BOD_5 concentration of 130 mg/l, and required effluent BOD_5 is 10 mg/l. (*Note*: These are the same conditions used in Example 1, so comparisons can be made.) The predominant wetland plant type in surrounding marshes is cattail.

Solution

1. Choose cattail for this SF system since it is successfully growing in local wetlands. From Table 1, cattail rhizomes penetrate about 30 cm into the medium. So the bed media depth (d) should be 30 cm.
2. Choose a slope of 1% ($S = 0.01$) for ease of construction.
3. Choose a gravelly sand as the medium. From Table 6, $k_s = 500\ \mathrm{m^3/m^2}$ day. Check the suitability of a 1% slope.

$$k_s S = (500\ \mathrm{m^3/m^2\ day})(0.01) = 5.0\ \mathrm{m/day} < 8.60\ \mathrm{m/day}$$

4. From Table 6, $k_{20} = 0.86$. Solve for K_T using Eq. (12).

$$K_T = k_{20}(1.06)^{(T-20)} = 0.86(1.06)^{(25-20)} = 1.34\ \mathrm{day}^{-1}$$

5. Determine the cross-sectional area of the bed using Eq. (13).

$$A_c = Q/(k_s S) = 760/5.0 = 152\ \mathrm{m^2}$$

6. Determine the bed width.

$$W = A_c/d = 152/0.3 = 507\ \mathrm{m}$$

7. Determine the surface area required using Eq. (12).

$$A = \frac{Q(\ln C_0 - \ln C_e)}{K_T d\eta} = \frac{(760\ \mathrm{m^3/day})(\ln 130 - \ln 10)}{(1.34)(0.3)(0.35)}$$

$$= 13{,}854\ \mathrm{m^2},\ \text{requiring more area than FWS constructed wetlands due to}$$

$\times$ low porosity.

8. Determine the bed length (L) and the retention time (t) in the system.

$$L = A_s/W = 13{,}854/507 = 27.3\ \mathrm{m}$$

$$t = LWd\eta/Q = (27.3)(507)(0.3)(0.35)/(760) = 1.91\ \text{days}$$

16.6. Other Considerations

16.6.1. Hydraulic Budget

For a CW, the water balance can be expressed as follows:

$$Q_i - Q_0 + P - \mathrm{ET} = \frac{\mathrm{d}V}{\mathrm{d}t} \tag{14}$$

where Q_i is the influent wastewater flow, volume/time; Q_0, effluent wastewater flow, volume/time; P, precipitation, volume/time; ET, evapotranspiration, volume/time; V, volume of water; t, time.

Groundwater inflow and infiltration are excluded from Eq. (14) because of the impermeable barrier. If the system operates at a relatively constant water depth ($\mathrm{d}V/\mathrm{d}t = 0$), the effluent flow rate can be estimated using Eq. (14). Historical climatic records can be used to estimate precipitation and ET.

16.6.2. Site Selection

A CW can be constructed almost anywhere. Because grading and excavating represent a major cost factor, topography is an important consideration in the selection of an appropriate site. In selecting a site for an FWS wetland, the most desirable soil permeability is 10^{-6}–10^{-7} m/sec. Sandy clays and silty clay loams can be suitable when compacted. In heavy soils, additions of peat moss or top soil will improve soil permeability and accelerate initial plant growth.

16.6.3. Flow Patterns

A CW cell is designed to use one or more of three types of flow patterns: plug flow, step feed, or recirculation. Plug flow (Fig. 3a) is once-through flow down the cell length. Plug flow is now used for most municipal and acid drainage systems and requires minimal piping, energy use, operation, and maintenance.

Step feed (Fig. 3b) may benefit pollutant removal by using more of the CW area for solids removal and by providing carbon for nitrogen removal in the lower bed area. Step feed is typically combined with recirculation. The cost–benefit of step feed needs investigation.

Recirculation (Fig. 3c) should be considered, and its potential needs further investigation. Recycling treated effluent will dilute influent BOD_5 and SS, decreasing odor potential and increasing dissolved oxygen concentration and retention time, which will enhance nitrification and subsequent nitrogen removal (Steiner & Freeman, 1989).

16.6.4. Slope

A slope of 0.5% or less, as limited by construction tolerances, is recommended for an FWS system. Some slope is needed to drain the cell for maintenance and possible mosquito control. SF bed slopes should be 2% or less based on the initial hydraulic conductivity of the substrate. A level substrate surface is recommended for an SF system to facilitate vegetation planting and to control weed by flooding the bed. Bed slopes greater than 4% would cause most of the flow to be on the surface due to poor hydraulic conductivity, with channeling, treatment, and vegetation management problems (Steiner & Freeman, 1989).

16.6.5. Liners

If groundwater contamination or water conservation is a concern, an impermeable liner below the substrate is required. Possible materials are compacted in situ soil

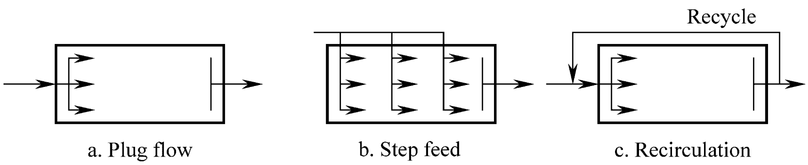

Figure 3: Flow patterns for constructed wetlands.

(permeability less than 10^{-6} cm/sec); bentonite; asphalt; synthetic butyl rubber; or plastic membranes. The liner must be strong, thick, and smooth to prevent root penetration and attachment (Steiner & Freeman, 1989).

16.7. Operation and Maintenance

Two operational considerations associated with wetlands for wastewater treatment are mosquito control and plant harvesting. In addition, system disturbances can occur from time to time.

16.7.1. Mosquito Control

Mosquito problems may occur when wetland treatment systems are overloaded organically and anaerobic. Strategies used to control mosquito populations include effective pretreatment to reduce total organic loading; step feeding of the influent wastewater stream with effective influent distribution and effluent recycle; vegetation management; natural controls, principally by mosquito fish (*Gambusia affinis*), in conjunction with the above techniques; and application of man-made control agents. In general, natural controls are preferred because of a concern that man-made control agents resistant might develop resistant strains of mosquito (Wieder et al., 1989).

16.7.2. Plant Harvesting

The usefulness of plant harvesting in wetland treatment systems depends on several factors, including climate, plant species, and the specific wastewater objectives. Harvesting plants to remove wastewater contaminants taken up by the plants is inefficient. However, plant harvesting can affect treatment performance of wetlands by altering the effect that plants have on the aquatic environment. Further, because harvesting reduces congestion at the water surface, control of mosquito larvae using fish is enhanced. Where a segmented wetland system is used, drying each segment separately allows harvesting with conventional equipment. Depending on location, burning the dried plant mass in place may be most economical (Wieder et al., 1989).

16.7.3. System Perturbations and Operation Modifications

Deviations from "average" operating conditions will occur in the system lifetime with greater frequency than predicted or preferred. Perturbations generally are of two types (Girt & Knight, 1989): (1) predictable perturbations, which can be predicted and occur periodically; and (2) unpredictable perturbations, which are unforeseen in the design phase or which occur so infrequently that incorporation into the design would entail unnecessary expense. Tables 7 and 8 summarize, respectively, predictable and unpredictable events, along with operational features most affected, associated symptoms, and appropriate operation modifications.

16.8. Case Studies

16.8.1. Case Study A: Emmitsburg, Maryland, USA, SF Constructed Wetland

The town of Emmitsburg constructed an SF system with a design flow rate of 130 m^3/day in 1984 to treat a portion of its effluent flow. The system is a single basin, 76.3 m long, 9.2 m wide, and 0.9 m deep, filled with 0.6 m of crushed rock. Clay was used in the bottom of the basin to prevent groundwater contamination. Perforated pipes placed near the bottom of the basin are used for influent distribution and effluent collection. The water level during normal operation is approximately 5 cm below the surface of the gravel. The system was initially seeded with 200 broadleaf cattail plants in August 1984 and another 200 in July 1995. By March 1986, about 35% of the basin surface area was covered by cattails. The planting density used should have been at least an order of magnitude higher. Until the plants cover the entire basin, the performance will not be representative of an SF system.

The influent to the Emmitsburg system is trickling filter effluent. Influent flows vary between 95 and 132 m^3/day, which corresponds to a surface hydraulic loading rate of 1,420–1,870 m^3/ha/day. Effluent samples are collected and analyzed weekly for BOD_5, SS, TDS, DO and pH.

The influent BOD_5 concentrations to the system range between 10 and 180 mg/l while SS concentrations normally range between 10 and 60 mg/l. Results from two years of operation are presented in Table 9, which show very good performance even with the limited plant coverage. Odors in the effluent have been an occasional problem but the frequency of noticeable odors is decreasing as cattail coverage increases. The engineering and construction costs of the Emmitsburg system were less than US$35,000.

16.8.2. Case Study B: The Eastern Seaboard Industrial Estate (ESIE), Rayong Provice, Eastern Thailand, Vertical-Flow Constructed Wetlands

The ESIE comprises of over 100 industrial factories and 14,000 staff. The industries are required to pre-treat their wastewaters to remove heavy metals

Table 7: Predictable system disturbances.

Disturbance	Features	Symptoms	Operation modifications
Startup	Difficulties in vegetation establishment and microbial flora colonization	Widely fluctuating treatment efficiencies	Control loading rates, i.e. water flow rate, chemical concentration
			water inflow rates
			freshwater source
			Control water depths — critical for vegetation establishment and development of conditions suitable for target microbial populations
			Dilution/recirculation
			Chemical additions (e.g. lime to improve soil pH, fertilizer)
Seasonal	Extreme precipitation	High loading rates	Control loading rates, i.e. water flow rates, chemical concentration
			Water inflow rates
			Dilution
			Recirculation
		Decreased storage capacity	Increase storage capacity
			Stormwater diversion
			Detention pond
			Increase water depth
		Insufficient residence times;	Control outflow rates
		Channeling	Install baffles
	Extreme low temperatures	Insufficient flow	Freshwater source, recirculation, parallel cells
		Freezing; sheet flow over ice surface	Recirculation, aeration, control water depth
			Preheated water

(*continued*)

Table 7: Continued.

Disturbance	Features	Symptoms	Operation modifications
	Vegetation growth/decay	Flushing of chemicals, nutrients, and microbes from decaying vegetation, sediment	Secondary treatment pond
			Vegetation harvest or burning Recirculation to increase nutrient retention
	Population composition changes (microbial, algal)	Gradual change in treatment efficiency	Secondary treatment pond
			Recirculation

Girt and Knight (1989).

and other toxic compounds according to the Thailand effluent standards prior to discharging into combined sewers and mixing with other domestic wastewaters. The current wastewater flow rate (year 2000) is about 7,000 m^3/day.

This combined wastewater is being treated by two CW in series, each with a dimension of $35 \times 18 \times 0.8$ m^3 (length $\times$ width $\times$ depth). The CW beds are lined with high-density polyethylene sheet and filled with 1-cm gravel. Wastewater is applied intermittently over the CW beds in a vertical-flow mode, and the percolates are collected through underdrainage pipes. Cattails, bulrushes and canna are the primary vegetation grown in these CW beds.

During the period of September 2001–May 2002, these CW units were operated at the following conditions: hydraulic loading rates 60–160 l/m^2/day; organic loading rate 57–140 kg BOD_5/ha/day and hydraulic retention times 1.4–4.0 days. The treatment performance of these CW units in series was found satisfactory (Table 10) with the effluent BOD_5 and SS concentrations being 4 and 10 mg/l, respectively. Because of the nitrification reactions occurring in the CW beds, there was an increase in NO_3-N concentration from 0.06 mg/l in the effluent to 4.75 mg/l in the effluent. This treated water is being sold to some factories located in the ESIE for uses in factory air-cooling and other processes.

Due to prolific growth of the vegetation under tropical conditions, plant harvesting was done once in 4 months, with annual yields of 130–150 t/ha/yr (wet

Table 8: Unpredictable system disturbances.

Disturbance	Symptoms	Operation modifications									
		Water inflow	Water outflow	Water depth	Dilution	Recirculation	Pretreatment pond	Chemical addition	Vegetation harvest	Re-plant	Predator control
Record storm event	High hydraulic loading rates	x				x	x				
	Decreased storage capacity	x					x				
	Insufficient residence time	x	x	x	x	x	x	x			
	Channeling	x					x				
	High sediment loads			x			x				
	High chemical loads		x		x	x	x	x			
Change in chemical constituents and concentrations	High chemical loads	x			x		x	x			
	Increased toxicity (vegetation, wildlife)				x			x	x	x	
	Release of chemicals from Sediments/ vegetation		x	x		x			x	x	
	Change in chemical form			x	x						

(*continued*)

Table 8: Continued.

Disturbance	Symptoms	Operation modifications									
		Water inflow	Water outflow	Water depth	Dilution	Recirculation	Pretreatment pond	Chemical addition	Vegetation harvest	Re-plant	Predator control
Vegetation damage	Increased debris, flow hindrance	x		x					x	x	
	Nutrient release from vegetation		x	x					x	x	
	Change in conditions for replanting	x		x		x	x		x	x[a]	
	Complaints from neighbors			x				x			x
Pests (rodents, mosquitoes, etc.)	Reduced flow and water level control	x	x	x	x	x	x	x			
	Inability to respond to need for changes in operations[b]										
Malfunctions/ construction failures	Limited treatment capacity[b]										
	Limited lifespan[b]										
Design flaw											

Girt and Knight (1989).
[a] New species.
[b] All operation modifications may need to be considered.

Table 9: Performance of the Emmitsburg SF system.

Season	Average flow (m^3/day)	BOD_5 (mg/l)		SS (mg/l)		Effluent DO (mg/l)	Area covered with cattails (%)
		Inf.	Eff.	Inf.	Eff.		
Fall 1984	117	29	12	25	7	1.0	<5
Winter 1985	111	68	29	37	9	0.3	<10
Spring 1985	130	117	38	37	13	0.0	<20
Summer 1985	100	87	11	28	10	1.3	<25
Fall 1985	97	28	7	29	7	2.1	<30
Winter 1986	106	40	11	25	4	–	<35

U.S. EPA (1988).

weight). These harvested plants can be used in making furniture and other decorations, another income generation for the ESIE.

16.8.3. Case Study C: Vertical-Flow Constructed Wetlands for Septage Dewatering and Stabilization, Asian Institute of Technology (AIT), Bangkok, Thailand

Septic tank sludge (or septage) usually contains high solid, organic and enteric microorganism contents, with poor setting and dewatering characteristics. A pilot

Table 10: Treatment performance of the CW unit.

Parameter	Average concentration (mg/l)			Overall removal (%)
	Influent	Effluent unit 1[a]	Effluent unit 2[b]	
BOD	88	19	4	95.8
COD	229	52	19	91.5
SS	98	16	10	89.5
TKN	24.1	14.5	4.6	81.1
NH_3-N	14.2	10.8	3.3	76.7
NO_3-N	0.06	0.53	4.75	–[c]
TP	7.0	4.7	1.5	78.5

AIT (2002).
[a] Percolate of CW unit 1.
[b] Percolate of CW unit 2 in series.
[c] Increased likely due to nitrification reaction.

study was carried out at the AIT campus during 1998–2001 to demonstrate the feasibility of applying vertical-flow CW to dewater and stabilize septage collected from Bangkok city, Thailand.

Three pilot-scale CW beds, each with a surface area of 25 m^2, having 65 cm sand–gravel substrata, supported by a ventilated drainage system (Fig. 4) were planted with narrow-leave cattails (*Typha augustifolia*). To operate in a vertical-flow mode, the Bangkok septage (Table 11) was uniformly distributed on the CW beds 1–2 times weekly at the solids loading rate (SLR) of 1.5–9.6 kg TS/m^2 week, while the percolate was collected from the drainage system for further treatment.

The experimental data obtained so far indicated the SLR of 4.8 kg TS/m^2 week to be the most suitable, resulting in the highest TS, COD and TKN removal of 80, 96 and 92%, respectively (Koottatep et al., 2001). The TS contents of the dewatered septage remaining on the CW beds were increased from 1–2 to 30–60% with the one-week operation cycle. Because of the vertical-flow mode of operation and with the effectiveness of the ventilation pipes, there was a high degree of nitrification occurring in the CW beds — the NO_3-N contents were found to increase from about 4 mg/l (in the Bangkok septage) to 180–250 mg/l in the CW percolate.

Due to rapid flow-through of the percolates, there was little liquid retained in the CW beds, causing the cattail plants to wilt, especially during the dry season. To reduce the wilting effects, the operating strategies in the second year were modified by ponding the percolate in the CW beds for periods of 2 and 6 days prior to discharge. This operating strategy was found beneficial not only for mitigating plant wilting, but also for increasing N removal through enhanced denitrification activities in the CW beds. During the 3-year operations, the dewatered septage was

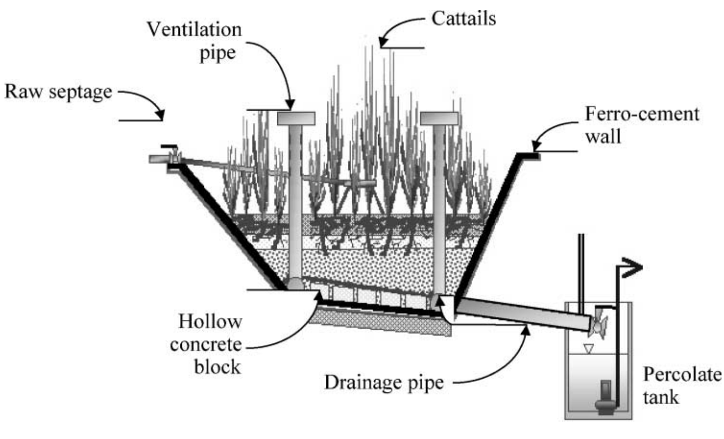

Figure 4: Schematic diagram of pilot-scale CW units.

Table 11: Characteristics of Bangkok septage samples.

Parameter	Range	Average	Standard deviation
pH	6.7–8.0	7.5	0.6
TS (mg/l)	2,200–67,200	19,000	12,500
TVS (mg/l)	900–52,500	13,500	9,400
SS (mg/l)	1,000–44,000	15,000	10,100
BOD (mg/l)	600–5,500	2,800	1,400
TCOD (mg/l)	1,200–76,000	17,000	15,000
TKN (mg/l)	300–5,000	1,000	800
NH_4 (mg/l)	120–1,200	350	170
NO_3 (mg/l)	1–11	4.5	3.5
Helminth eggs (no./g of sample)	0–14	4	1

Based on 120 raw septage samples during August 1997–February 1999.

not removed from the CW beds and no adverse effects on the septage dewatering efficiency were observed.

As can be expected, the dewatered spetage, after 2–3 years of decomposition in the CW beds, was a well-stabilized sludge without the presence of viable helminth eggs, suitable to be used as a fertilizer or soil conditioner.

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Chapter 17

Planting, Selection and Plant Establishment in Constructed Wetlands in a Tropical Environment

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Abstract. This chapter examines plant selection, sourcing, planting and establishment in constructed wetlands. The chapter deals with surface water constructed wetlands with water quality treatment as the primary objective. However, much of the information can be readily applied to other constructed wetland types. Information presented is based on the author's experience with constructed wetland projects in Australia and South-East Asia. The aim is to provide information relevant to the establishment of a desirable crop of healthy wetland plants within constructed wetlands.

17.1. Introduction

Frequently the establishment of wetland plants in constructed wetlands is perceived to be relatively straight forward given the lushness of vegetation growth in tropical environments. Indeed many desirable wetland plants are considered to be problematic weeds (Sainty & Beharrell, 1998c; Tjitrosemito, 1993). However, in practice this may not be the case. Scientific literature is limited in terms of plant selection and plant establishment in constructed wetlands not only in the tropics but all regions. This chapter is based on the authors experience with surface water constructed wetlands throughout Australian and in regions of South East Asia.

Good decisions concerning plant selection and incorporation of plant requirements into the design, can prevent a wetland from being an engineering success, but a biological failure. Even rudimentary knowledge of the issues involved in planting, selection and plant establishment can enhance the development of a dense stand of desirable species. Performance of the wetland relies not only on good design, but also on good construction and operation. Healthy wetland plants are a key feature affecting the consistent performance of wetland treatment systems (Kadlec & Knight, 1996a; Adcock et al., 2000).

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17.1.1. What is a Constructed Wetland?

Constructed wetlands can be considered as wetlands built to fulfill desired objectives. White (1998) defines a constructed wetland as "*purpose built structures, utilizing the predominantly natural materials of soil water and biota, which perform the desired physical, chemical and biological processes and functions of natural wetlands to achieve desired objectives*."

The objectives of constructed wetlands will include one or more of the following (White et al., 1996):

1. water quality improvement, including municipal waste, urban stormwater, agricultural and urban run-off, industrial and mine wastewater;
2. modify water flow;
3. provision of habitat;
4. passive recreation and visual amenity;
5. public education; and
6. modify water flow.

There are also several types of constructed wetlands these include: surface water wetlands; subsurface treatment wetlands; and natural treatment wetlands (Kadlec & Knight, 1996b). This chapter looks at plant selection sourcing and establishment in surface water wetland, where the primary objective is water quality treatment.

17.1.2. Roles of Water Plants in Constructed Wetlands

Wetland plants are an important component in constructed wetlands. The roles that they can fulfill or to which they can contribute to are numerous (Gersberg et al., 1986; Mitsch & Gosselink, 1993; Chambers et al., 1995). In a surface water constructed wetland the roles of wetland plants can include:

- aiding in the biochemical processes, which reduce nutrient and other pollutant concentrations;
- influencing sediment deposition and physically filtering sediment particles from the water column;
- influencing hydrology and hydraulics in constructed wetlands by increasing flow roughness and transpiration;
- providing shade, thus decreasing light availability for algal photosynthesis;
- decreasing erosion by reducing wave energy and flow velocities while binding soil particles with their root systems;
- providing a basis for wetland food chains and supplying shelter for invertebrates, amphibians, reptiles, birds and mammals; and

- improving visual amenity by adding color, texture, contrast and variety of patterns in the landscape.

17.1.3. What is a Wetland Plant?

Wetland plants have adaptations that allow the transportation of oxygen to the roots and rhizomes (Brix, 1997). This oxygen is needed not only for respiration in saturated soils by root or rhizome, but also for the leakage of oxygen which prevents toxins from accumulating in the root zone under saturated conditions.

Although there are numerous definitions of a wetland plant, none are satisfactory as there are always exceptions that fall outside the definitions. However, plants that thrive and flower in soil that is saturated for long periods can be considered wetland plants (Sainty & Beharrell, 1998a). There are notable exceptions such as *Phragmites australis* (Common Reed), which will grow in damp pasture land where soil is not saturated.

Generally, wetland plants can be divided into four groups:

(i) *Littoral plants* These species thrive in periodically flooded land and usually grow amongst species that are not strictly wetland plants. Notable species in this group are some *Juncus* spp. (Rushes) and *Cyperus* spp. (Sedges).
(ii) *Emergent plants* Emergent plants contain most of the wetland plant species; these species grow through the water column, examples include *Eleocharis* spp. (Spikerushs) and *Scirpus* spp.
(iii) *Submerged plants* Submerged plants are rooted or free-floating plants with their foliage entirely below the water surface. Examples include *Potamoge-ton* spp. (Pondweeds) and *Vallasenairia* spp.
(iv) *Floating plants* These float at the water surface. This type includes the Duckweeds *Lemna* spp, *Spirodela* spp. and, *Azolla* spp. and the introduced noxious weeds *Salvinia molesta* and *Eichhornia crassipes* (Water Hyacinth). It also includes the floating attached plants such as *Nymphoides* spp. (Marshworts) and *Nymphaea* spp. (Water Lilies).

17.1.4. Plant Protection

Pre-planning to ensure planting success is necessary at the design stage of any wetland project. The success of any surface water constructed wetland will be reflected in addressing issues associated with the catchment (Sainty & Dalby-Ball, 2000). Some of which are listed below:

- Will plants be dislodged or washed away. To prevent this wetlands should be built off-line or if on-line built to appropriate sizing to dissipate velocities.

- Sediment and gross pollutants must be extracted prior to flows entering vegetated areas of the wetland. Deposition of sediment can alter surface levels within the wetland affecting wetland performance, and smother plants.
- What potential pest animals can occur in the catchment? Pests such as the *Pomacea* sp (Pink Snail; Lim et al., 1999) found in areas of South East Asia has the potential to significantly affect plantings. Waterfowl can be a significant factor, in Sydney a group of Purple Swamphens (*Porphyrio porphyrio*) are known to have decimate 10,000 seedlings in one night.
- What pest species occur in the catchment? *S. molesta* and *E. crassipes* in the upper catchment can not only lead to long-term maintenance problems but can damage plantings in high flow events.
- Are there any pollutants that may harm plants? Submerged planting, e.g. may be damaged by elevated levels of chlorine or high turbidity (Sainty & Dalby-Ball, 2000).

When designing wetlands thought must be given to planting issues. Issues include: water depth and its control; flows and velocities; and expected performance of the wetland. For example some plant species can tolerate velocities up to 2 m/s, or recover rapidly after being knocked down. However, peak velocities should be kept below 0.5 m/s, with treatment process being limited when flows exceed 0.1 m/s (Wiese, 1998).

17.2. Wetland Plant Selection

Plant species and diversity should be selected to match the wetland objectives. For instance, where wetlands are restored or constructed primarily for conservation reasons, plant diversity will be required. On the other hand, a monoculture may be appropriate where wetlands are constructed to assimilate wastewater. A wetland planned as a wading bird habitat may require limited or low growing vegetation to encourage roosting or nesting (Saitilan, 2002).

Species diversity provides a variety of plant growth forms, with various habitat types and food for fauna. Diversity also provides protection against plant failure, which can be caused by some plant specific diseases or pests.

Structural diversity of wetland plants can be important for other reasons. This creates a larger range of predator species able to prey on mosquitoes (Russel & Kuginis, 1998).

17.2.1. What are We Looking for in Wetland Plant?

Constructed wetlands are a relatively new field in South-East Asia and as such there is limited information on their use. In many cases the choice of plants has been extrapolated from the same or similar plant species utilized elsewhere (Norazmi et al., 1999).

There are few studies on contaminant removal by wetland plants in the tropics. One investigation in Malaysia showed that all the species utilized had the potential for use in treatment wetlands. *Eleocharis dulcis* (Water Chestnut) produced the highest removal rates compared to the other species for a variety of available nutrients (Sahidin et al., 2001).

Laboratory removal rates results do not always transfer readily to field situations. This is due to the large number of variables found within constructed wetlands, particularly hydraulic and pollutant loadings. Pollutant removal rates and other characteristics of a particular species can vary on a regional basis due to genetic variability.

Most wetland species in South East Asia may play a small part in increasing diversity in a constructed wetland, but are especially suited to planting in constructed wetlands. Table 1 lists wetland plant species that have potential for use in constructed wetlands, they are found in many tropical environments in South-East Asia and Australasia. This list is by no means exclusive, and wherever constructed wetlands are proposed catchment investigations must be undertaken in order to decide which are the most appropriate species.

Generally, in surface water treatment wetlands emergent species are chosen. There are several basic requirements for wetland plants in a surface water constructed wetland. These characteristic include:

- being locally native species is an important criterion because these will be adapted to local conditions, and therefore have less potential impact on the local environment than introduced species, which are unpredictable and may cause significant environmental harm.
- having the ability to tolerate saturated soils and persistence in permanently flooded conditions.
- where possible being perennial rather than annuals.
- being robust plants which are more likely to withstand or recover from high flow events, they should also be able to tolerate variations in water depth.
- be able to produce leaf litter as sites for physiochemical activity, this is important in the processes involved in nutrient reduction with constructed wetlands (Broderick et al., 1988);
- being readily propagated; and
- having minimal weed potential, even locally native species can become significant weeds under certain circumstances.

Table 1: Some potential wetland plants for surface water constructed wetlands in the tropics.

Wetland plants	**Planting depths**[a]
Cyperus compactus	Littoral
Cyperus digitatus	Littoral
Cyperus halpan	Littoral
Eriocaulon longifolium	Littoral
Eleocharis dulcis	0.6–0.9 m
Eleocharis variegata	0.3–0.6 m
Fimbristylis globulosa	0–0.3 m
Fimbristylis miliacea	Littoral
Fuirena umbellata	0–0.3 m
Hanguana malayana	0–0.3 m
Lepironia articulata	0.3–0.6 m
Ludwigia adscendens	0–0.3 m
Ludwigia octovalvis	Littoral
Monochoria hastata	0–0.3 m
Nymphaea nouchali	0–0.6 m
Pandanus immersus	0–0.3 m
Phragmities karka	0–0.6 m
Phragmites australis	0–0.5 m
Phylidrum lanuginosum	0–0.6 m
Polygonum barbatum	0–0.3 m
Rynchospora corymbosa	0–0.3 m
Saccharam spontaneum	Littoral
Scirpus grossus	0–0.6 m
Scirpus juncoides	Littoral
Scirpus mucronatus	0–0.6 m
Scleria sumatrensis	0–0.3 m
Typha angustifolia	0–0.6 m
Vanda hookeriana	0–0.3 m

[a] The water depths can be variable and some species may tolerate greater depths with regular water level draw downs. Littoral refers to that area frequently inundated between operating water level to surcharge level.

17.2.2. Wetland Water Depths

It is important to note that there are distinct zones within a wetland in terms of water depth. Plant selection must be based on a design to ensure that they are suited to the hydrology of the new wetland. For example from the average operating water level to the surcharge level, inundation will be relatively frequent, but of a short duration. Dry spells will also be frequent.

Matching the depth of the wetland with appropriate plant tolerances is important. Different plants tolerate different water depths. With greater depth, diversity of species will be reduced, because the majority of emergent species are discouraged by depths greater than 0.6 m. Water quality in surface water treatment wetlands and genetic variability may significantly reduce a species tolerance to water depths observed in the wild (Kadlec & Knight, 1996c).

In surface water wetlands viability of many species at depths that they would normally tolerate is diminished if water depth remains consistent. Therefore, a lowering of water depth can significantly increase the viability of many emergent species. The logical time to reduce the depth is during the dry season when inflows are reduced. Currently, there is limited information on the duration and management of this water regime for wetlands in tropical areas. Lowering of water levels may also affect the ability of surface water wetlands to meet water quality objectives and may impact on habitat requirements.

17.2.3. Additional Species

Where diversity of plant species is required, additional species can be introduced. These include wetland plants that do not make up the bulk of the plantings, which can be introduced readily by use of propagules, e.g. *Ipomea aquatica* (Morning Glory) and *Ludwigia adscendens* (Water Primrose).

There are several native, decorative, flowering plants that can be used in constructed wetlands. Emergent species include *Philydrum lanuginosum* (Frogsmouth) and *Monochoria hastate* and floating attached species such as *Nymphoides* spp. (Marshwort), *Nymphaea* spp. (Water Lily), Lotus (*Nelumbo* spp.) and others which have attractive flowers. However, care in placement of floating attached plants must be made to ensure objectives are not compromised. Floating and floating attached species may invade open water and vegetated areas as well as blocking structures.

Deep sections of wetlands may benefit from the addition of submerged perennial plants. In Australia, these have been shown to contain the highest macroinvertebrate diversity in treatment wetland (Ross et al., 1997). However, care must be taken to ensure that other problems are minimized, these include blockages of structures and detachment which can lead to odor problems.

In surface water wetlands floating plants such as the noxious weeds including *S. molesta*, *E. crassipes* (Water Hyacinth) and *Pistia stratiodes* (Water Lettuce) are not desirable. In eutrophic conditions they can form a dense mat across the surface of then water. This prevents reoxygenation of the water column reducing water quality through anoxia. The mat also excludes light from the water column, reducing the diversity of fauna and the degree of pathogen disinfection by sunlight. They can also provide habitat for *Mansonia* spp. mosquitoes (Russell, 2000).

17.2.4. Trees in and Around the Wetland

Many trees have potential to be grown in shallow wetlands provided there is an annual total draining of free water from the site. There are many trees native to the tropics tolerant of shallow flooding including the Paperbarks (*Melaleuca* spp.) and She Oaks (*Casuarina* spp.). These may be added to a wetland containing emergent wetland plants (Sainty & Beharrell, 1998a).

Careful attention must be given to the selection and placement of trees near the wetland, as shading may hinder the growth and density of wetland plants (Beharrell, 1996). Trees may need to be positioned away from the edge of the wetland to a distance equal to the maximum potential height of that species. That is, a tree with a potential height of 20 m should be planted 20 m from the edge of the wetland.

17.3. Sourcing Plant Material

The propagation and supply of plant material is a booming business in the USA, Great Britain and parts of Europe (Kadlec & Knight, 1996a–c) and has rapidly expanded in Australia over that last 5 years.

Several sources and types of plant material are available for planting within the wetland and are discussed below. Certain species may be available in a particular form, e.g. in Australia, seed of *Bolboshenous fluviatalis* is rarely viable, bulbs are used instead. At any one site several types of material may be required. In the Putra Jaya project in Malaysia a mix of plant materials were used to provide the diversity and the numbers of plants required.

17.3.1. Direct Seeding

In some situations direct seeding can be a quick and cheap method of establishing vegetation. However, it requires careful management and a good knowledge of the germination and growth requirements of the species being used. Generally, direct seeding is not a viable option for establishing the majority of vegetation in constructed wetlands due to seed and germinated seedlings being at constant risk from:

- flooding;
- predation by water fowl, insects etc.;

- desiccation;
- fungal attack; and
- weed invasion.

17.3.2. Translocation

This is another form of direct seeding utilizing using the whole seed bank. The soil containing a viable seed bank is removed in layers and translocated to a suitable alternative site (Kadlec & Knight, 1996a; Brock, 1997). This method has been successfully utilized when a wetland site is to be developed and an appropriate translocation site exists. However, similar problems to direct seeding exist and precautions must be taken to ensure that major pests/diseases are not introduced.

17.3.3. Transplanting

Transplanting is a process of removing plants or plant pieces from existing wetland area. Where possible, disturbance of natural wetland areas should be avoided. Plant pieces should only be taken from existing wet areas only when it has been established that removal will not damage wildlife habitats. Where diverse and large numbers of plants are required, identifying the potential areas for sourcing has to be done early.

When it is undertaken, this technique for establishing vegetation has the advantage that the transplanted stock will be relatively mature. Rhizomes or tussocks are collected by digging up existing vegetation. This method can be particularly successful for *Phragmites* spp. (Reed) and *Scripus* spp. (Clubrushs). Many species, however, will not transplant well if the leaves and stems are damaged. Root pieces must be managed carefully after extraction and kept cool.

Alternatively, harvested vegetation may be used as a propagation material for example cuttings may be taken. Propagation of *P. karka* and *P. australis* by this method has been shown to be particularly successful in Malaysia and Australia respectively.

Potential problems with transplanting including:

- the fact that only areas with large populations of the plants can be utilized and there is the potential for significant impact on the natural environment;
- the possibility of importing noxious weed species;
- that many wetland plants are large and difficult to handle;
- that large scale harvesting requires stockpiling, leading to double handling and difficulty in maintaining viability during storage;

- harvesting and transplanting, which is very labour intensive, and may require plants to be transported over significant distances; and
- ensuring safety and security during field collection (e.g. use of life jackets, protective boots, first-aid kits and protection against human pests and predators).

17.3.4. Nursery Propagation

Nursery grown seedlings can be produced in high volumes, of even age and quality, that can be planted at any density. Nursery stock has several advantages, including:

- seedlings that are generally of a uniform height and age;
- that the maturity and quality of the plant stock can be pre-determined; and
- that accidental introduction of weed species can be minimized.

Some wetland plants are easily propagated and it may be feasible to propagate plants on-site. Propagation of nursery seedlings is relatively simple. However, a significant lead time may be required to allow for seed collection and development of propagation techniques.

Propagation by tissue culture is not recommended. Wetlands developed with plants sourced by tissue culture are derived from one or few genotypes and lack the genetic diversity obtained from using seed. Such uniform plant stock is more susceptible to predators, disease and climatic extremes (Sainty & Beharrell, 1998a).

Before purchasing seedlings for the project, it is recommended that a written guarantee is obtained stating that the seed was collected from an agreed location, and that the plant stock is weed free.

Types of Seedlings. Seedlings can be propagated for a variety of plant materials. These include seeds, corms, bulbs, cuttings, and division of tussocks. The viability of these propagules is highly variable and can be dependent on genotype and climatic conditions.

Seedlings can be made available for nurseries in different sized containers reflecting the maturity of the plant stock.

Small cells (20–100 ml volume) may be suitable for some species where there is:

- a well-prepared and level substrate;
- good control over water levels; and
- the wetland is off-stream (i.e. seedlings will not be decimated in high flow events).

Generally, larger cells with a volume over 200 ml are more robust and will have a higher rate of success. While seedling size is important, there are variations between species. For some species, increasing the size of the seedlings purchased and decreasing the planting density may be appropriate. The more mature and healthy the seedling, the greater the chance of survival. All seedlings must be "Hardened-off" prior to planting to minimize any planting trauma.

Plants Strips. Frequently used in USA and Australia are strips of mature plants specially grown and subsequently harvested. This provides a mature plant product similar to grass turf, which can be readily laid within the wetland. Whilst the cost is significantly greater it provides an instant result and can be used to establish vegetation at a quicker rate.

17.3.5. Planting Design

A planting design is a plan showing placement within a wetland. A planting design will allow the area for each plant species to be pre-determined. Numbers of each species required can then be estimated using the plant densities for that species. The planting design will make management of planting simpler. Areas and densities should be shown to scale on the design.

Surface water treatment wetlands are often planted with emergent species in bands at right angles to flow. Water flowing into the wetland then passes through a uniform vegetated band reducing the possibility of water finding the path of least resistance (i.e. short circuiting). It is important to differentiate littoral vegetation from other water plants. Littoral species can tolerate inundation and/or shallow depths (up to 200 mm) but are also adapted to extended dry periods. These species should be planted in a littoral zone where intermittent inundation will occur. In the reed bed zone, emergent vegetation adapted to deeper and more permanent water levels is required.

If continued diversity is an objective rapidly spreading rhizomatous wetland plants are best planted next to strong competitors. Thus the rapidly spreading *P. karka* should not be planted next to slow growing *Eleocharis* spp. (Spikerush). *Phragmites* spp. (Reed) are aggressive, fast-spreading clonal species which have demonstrated the capacity to dominate a wetland over time reducing diversity. Growth rate, speed of spread and ultimate vigor are all factors in deciding where to position wetland plants.

17.3.6. Planting Density

Planting densities vary from 1 plant/5 m^2 for *Melaleuca* spp. (Paperbark) to 8 plants/m^2 for some emergent plants. Densities depend on:

- the plant material used;
- growth rates of species; and
- the nature of the root system.

Rhizomatous species such as *S. grossus* and *P. karka* are capable of spreading many meters in a few months and 6 plant/m^2 may be sufficient. Tufted species such as *S. mucronata* may require a density of at least 10 plants/m^2. Densities must be high enough to crowd out weed species.

In the Putra Jaya Project in Malaysia planting densities of 12 plants/m^2 for large seedlings have been used with outstanding success. In Australia densities as low as 4 plants/m^2 for similar plant material have been used successfully where ideal plant establishment conditions existed.

Planting density has a significant influence on the cost of establishing a wetland. It can also be influenced by other factors including:

- time of planting (i.e. season);
- quality of substrate preparation;
- ability to provide adequate establishment conditions; and
- plant material used.

17.3.7. Seed Collection

For seed collection, timing is important to ensure that the fruits/seeds are mature at collection. If seeds have to be collected, requests for supply should be made a minimum of 12 months prior to planting, as different species may set seed at different times of the year. If propagation is delayed there is potential for losses due to using immature plants.

17.4. Planting

Generally, the type of plant material used determines planting techniques. The key to successful planting is to minimize the transplant shock during planting.

17.4.1. Planting Timeframes

The planting and establishment of wetland vegetation is crucial for the success of constructed wetlands. Depending on wetland objectives and planting techniques, planting may not be required, or proponents may rely on natural revegetation and colonization for plant establishment. However, such situations are not common.

Planting is dependent on wetland construction being completed and so it is important to determine construction and subsequent planting timeframes. Planting should occur as soon after preparation of the wetland substrate as possible to reduce the chance of the substrate being either eroded or invaded by weeds.

There are several factors that affect the establishment of a dense cover of vegetation in a constructed wetland, including:

1. correct placement of various species within the wetland;
2. quality of plant material;
3. preparation of the planting substrate;
4. planting techniques minimizing shock to the transplanted stock; and
5. provision of ideal conditions for plant establishment.

17.4.2. Planting Substrate

Within constructed wetlands a planting substrate to support plant growth is provided on the subsoil exposed by excavation. This exposed subsoil will usually be too compacted to allow plant root growth and may also lack nutrients. However, on occasions planting may be undertaken on undisturbed sites with no substrate. In these instances it will be necessary to control the existing vegetation.

In subsurface flow wetlands a specific substrate will be required to support plant growth whilst allowing water movement.

17.4.3. Selection of Substrate

The provision of an appropriate substrate can be the key to successful plant establishment (Norazmi et al., 1999). The selection of a suitable substrate is based on horticultural principles, i.e. plants need support, ability for roots to grow downwards and nutrients. Friability for root growth is very important dense soils such as clays will prevent root growth. Soils with a clay content greater than 20% may be problematic.

For a surface water system the minimum depth of any substrate should be 250 mm. It is more convenient and less costly if substrates can be used from the wetland construction site. This should be identified in site investigations. In such situations, the substrate material should be carefully stockpiled and protected against erosion for later placement in the wetland.

Substrate materials should be physically and chemically suited to their intended location within the wetland. For instance, highly erodable materials should not be chosen for high energy areas of wetlands. Similarly, wetland substrates should be inherently chemically stable, e.g. non-dispersive (Sainty & Beharrell, 1998b).

Fertilizer may be used where the substrate is poor and devoid of nutrients: in Australia soils with total available phosphorus less than 5 mgP/kg are considered nutrient poor. A controlled-release fertilizer can be used (Kuginis et al., 1998), however, slow release fertilizer must be specifically designed to tolerate the continuous wet conditions (Jacobs, 2000).

Substrates may also contain many weed seeds, especially those derived from other wetland areas, floodplains, riverbeds, etc. The use of substrate material known to come from an area with weed populations should be avoided.

17.4.4. Preparation of Substrate

If substrate preparation has been poor and levels are uneven, the wetland will not drain and maintenance costs may be increased or mosquito habitat enhanced. Maintaining friability and accurate leveling are the most important aspects of substrate preparation.

The substrate needs to be placed and leveled in the wetland. Any substrate ameliorant (fertilizers, gypsum, lime, organic matter, etc.) can be tyned in after placement. It is important to provide a level planting bed, but ensure that the substrate is not overworked with machinery, as this leads to a declines in substrate structure. The substrate should not be compacted.

Some substrates may be left prepared for some time before ideal conditions for planting arise. This should pose few problems as long as substrates are protected from erosion and any weed growth is treated.

17.4.5. Planting Methods for Wetlands

Ideally, plant into damp or dry soil and irrigate after planting. Planting into wet mud or shallow free water is also possible, and depending on the circumstance may be the only practical way, but it is more time consuming.

Mechanical planting techniques are being developed and occasionally used in Australia and elsewhere to increase planting efficiency especially over large areas. Mechanical planting is restricted to planting into a dry substrate using relatively low plant densities, and may require the use of specific plant stock. Mechanical planting may damage a level substrate and plant roots.

Manual planting is the most widely used method. Manual planting allows appropriate densities, as required by the planting plan, to be realized. Manual planting can reduce damage to plant stock and lead to high plant survival rates. If the substrate is wet or is covered by water, manual planting may be the only option available.

All members of planting groups should be briefed about the objectives of the wetland and the requirements of the planting plan prior to planting. This session can also be used to explain and demonstrate the skills and tasks for planting. Other tasks may be allocated to various team members, for example watering plants and carrying plants to those doing the planting.

At the Putra Jaya wetland project in Malaysia, laborers were able to plant approximately 800–1,000 plants per day. In Australia individuals in professional planting teams, on a well prepared substrate, can average between 1,000 and 1,200 plants per day.

17.4.6. Supervision

Experienced personnel must always supervise the work. The 'planting manager' should understand the planting design and have a detailed knowledge of planting techniques for wetland species. This person should also be responsible for assigning people to the various tasks. Unless mistakes are promptly corrected, there will be a repetition of errors. The most common mistake in planting is for enthusiastic planters to plant at very high densities or use the wrong species.

The location and density of species planted in the wetland will often change from that set out on the planting design. Experience and common sense will allow decisions to be made regarding changes from the original planting design.

17.4.7. Irrigation

In dry conditions, seedlings need to be "watered in" within a few hours of planting. Subsequent irrigation will vary according to each site. If planting is to take several days or weeks, seedlings will need to be irrigated frequently.

An alternative is to flood cells a section at a time by using berms to separate unplanted areas from planted areas. Once all areas of the wetland are planted the banks can be broken. These low banks may be covered by water at normal operational water levels.

Occupational health and safety should be an essential part of all planting activities. If the wetland receives a heavily polluted water supply, ensure that there is no free water in the wetland so that possible health risks from pathogens during planting are prevented.

17.4.8. Plant Handling

Plant stock needs to be stored in a safe, secure location. If this location is not at the wetland site, stock should be taken to the site as required. Stored seedlings must be kept out of the sun and watered regularly. All plant stock should be labeled with the plant name and number of plants.

17.5. Plant Establishment

Many project managers consider the hardest part of the project over on the completion of planting. However, if conditions to encourage plant growth are not provided, plant deaths can be expected. Plant establishment is dependent on water level control and nurturing seedlings. Inspections should be undertaken at least weekly during this stage to ensure that actions which promote growth are undertaken.

17.5.1. Managing Water Levels After Planting

One significant impact of tropical environments is that precipitation exceeds evaporation. This means that correct water level management is crucial for plant establishment and maintenance.

During plant establishment, water levels within the wetland will need to be controlled to prevent seedlings from being desiccated due to lack of water, or drowned by excessive water levels. Initially, after planting water depths should be kept low. As a rule of thumb only one third of the shortest plant should be inundated. A common misconception of wetland plant growth is that they are dependent on elevated water levels, whereas in the majority of cases the opposite is true.

17.5.2. Weed Invasion and Plant Health

During plant establishment the wetland should be checked regularly for plant health and weed invasion. Weed invasion is a potential hazard during plant establishment as there is little competition from desirable vegetation and a dense cover has yet to be established. Weekly inspection can reduce the potential for minor infestations becoming major problems, displacing desirable vegetation.

Plant health may suffer during plant establishment due to transplant shock, disease, pest species and/or inappropriate water levels. Regular inspections of plant health can allow steps to be taken to rectify problems quickly.

17.5.3. Supplementary Planting

In all projects there will be areas where plant survival rates are poor. Contingency funds must be made available to re-plant areas with high plant mortality. Areas devoid of plants may be susceptible to weed invasion and flow short-circuiting. Knowledge of the most successful plantings at initial planting can be utilized to select replacement species.

17.5.4. Records

Details of all plantings should be formally kept. This will enable changes in planting density and species composition to be noted, and informed decisions can then be made about future re-planting and water level management. If planting in blocks, permanent markers should be installed for monitoring purposes. This will allow the success of the planting and changes in species composition to be assessed over time.

17.6. Conclusion

The establishment vegetation in constructed wetlands is dependent on attention to detail. There are many variables that need to be considered when establishing vegetation. Planning and design must take into account plant requirements specifically water depth and velocity. Potential pests, weeds and contaminants in the catchment likely to affect plant growth must be identified.

There are many plants that show potential for use in constructed wetlands in tropical climates. However, there is minimal knowledge on the propagation and requirements of these species. This is further complicated by there being likely variation based on genetic variability on a regional basis. However, it is important to ensure an appropriate water depth for the species used.

Successful growth of wetland plants is heavily dependent on the substrate of the wetland and water level control during establishment. These variables must be management successfully if a wetland is to succeed. It is during this phase that the majority of wetland plantings fail.

Wetland plants play an important role in the success of constructed wetlands. The establishment of a dense stand of desirable vegetation in a constructed wetland does not appear difficult. Currently there is limited information on plant sourcing, selection and establishment in tropical climates. With the increased use of constructed wetlands in the region the availability and development of appropriate techniques is sure to increase.

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Chapter 18

Nitrogen Removal Processes in Constructed Wetlands

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Abstract. Constructed wetlands can effectively remove nitrogen from waste-waters and diffuse run-off from land. In mature wetlands the dominant N removal process is generally microbial denitrification. N removal from ammonium-rich wastewaters is frequently limited by insufficient oxygen for initial nitrification. Alternative microbial N removal processes, including anaerobic ammonium oxidation, may be important in these situations. Emergent plants enhance N removal mainly via indirect effects on physico-chemical and microbial processes. In particular, they promote settling and retention of suspended solids, transport oxygen into the root-zone, provide surfaces for biofilm growth, and produce organic substrates for denitrifying bacteria.

18.1. Introduction

Constructed wetlands attempt to replicate and optimise treatment processes that occur naturally in swamps, fens and marshes. Efficiency is enhanced by optimising dispersion, flow paths, water depths, residence times, and vegetation characteristics. Constructed wetlands are now widely used to provide "natural" ecotechnological treatment solutions for urban, industrial and agricultural waste-, storm-and drainage-waters (USEPA, 1993, 2000; Kadlec & Knight, 1996; IWA, 2000). Construction and operating costs are low relative to mechanical treatment plants providing suitable land is available, and provision of wildlife habitat and green spaces may provide ancillary benefits. Much of the historical development and application of constructed wetlands has occurred in North America, Europe and Australasia, but interest is now rapidly increasing in Asia, South America and Africa.

Constructed wetland designs can be most simply classified as surface-flow (SF, also known as free-water surface) or subsurface-flow (SSF, also known as vegetated submerged beds, reed-beds and root-zone systems). In SF wetlands,

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the wastewater flows through a shallow "pond" planted with emergent plants such as bulrushes, reeds or sedges, or less commonly, floating or submerged macrophytes. In SSF systems, the wastewaters flow through gravel or similar substrata, and the plants grow rooted in the gravel. SF wetlands have become favoured in many areas of the world because they are cheaper to construct (no gravel media required) and generally have higher wildlife habitat values. SSF wetlands, however, tend to be more effective at suspended solids removal and BOD reduction per unit land area. Because the wastewater remains below the surface in these systems, there is also little possibility for human or wildlife contact with wastewaters and less potential for odours or insect infestation. Intermittently dosed, vertical-flow (VF) constructed wetlands have recently been developed to provide enhanced removal of biochemical oxygen demand (BOD) and nitrogen (IWA, 2000). These wetlands are essentially simple percolating filters with plants and will not be covered further here. The use of hybrid designs incorporating VF, SSF and/or SF sections is becoming increasingly common.

Key features of wetlands that contribute to their nutrient and contaminant removal functions include:

1. Low flow velocities and tortuous pathways through aquatic vegetation, which favour sedimentation and accumulation of particulates. BOD, nutrients and other contaminants associated with settled particulates are thus removed from through-flowing waters and incorporated into the wetland sediments.
2. Intimate contact between water, sediments, plants, detritus, and biofilm, which enhances assimilation of nutrients and substrates, and promotes physical, chemical and biotic interactions. Nutrients taken up by plants are recycled both internally within plants and through leaching and mineralisation of standing and fallen litter. A proportion of nutrients are bound up in detritus and refractory humic compounds which tend to accrete in the wetland. Organic substrates exuded by plants and released from decaying plant tissues can fuel important microbial transformations such as denitrification.
3. A mosaic of aerobic and anaerobic micro-environments, which promotes sequential microbial transformations of a wide range of nutrients, metals, and natural organic and xenobiotic compounds. High loadings of organic substrates and restricted oxygen exchange with the atmosphere create anaerobic conditions, particularly in the sediments of wetlands. Algal and submerged plant photosynthesis during daylight promotes aerobic conditions within the water column and in biofilms. Atmospheric gas exchange across the water surface via diffusion, convection, and release from the internal tissue of plants produce aerobic micro-zones near the water surface, and associated with shoots and the root-zone. In combination these opposing processes of oxygen

consumption and supply create a complex temporal and spatial mosaic of aerobic, anoxic and anaerobic environments in wetlands.

Nitrogen is an important contaminant in many waste, storm, and drainage-waters. Key forms of N in water include the oxidised species nitrate (NO_3) and nitrite (NO_2), and reduced species such as ammoniacal-N (NH_4-N) and N bound in dissolved and particulate organic matter (Org-N). Ammoniacal-N is a major plant nutrient that can promote excessive growth of aquatic plants, leading to eutrophication of water bodies. The un-ionised ammonia (NH_3^+) component (favoured at elevated pH and temperature) is also toxic to aquatic life and may exert a significant nitrogenous oxygen demand (NOD) as a result of bacterial nitrification processes (see below). This chapter introduces the key processes important for nitrogen removal in constructed wetlands, and then uses examples, from work of both the author and others, to illustrate the role of these processes in wetlands constructed for treatment of wastewaters and agricultural drainage waters.

18.2. Microbial Nitrogen Transformation Processes

Cycling of N in wetlands is complex, and includes important microbially mediated transformations (Fig. 1). Biological N-fixation is likely to be negligible compared

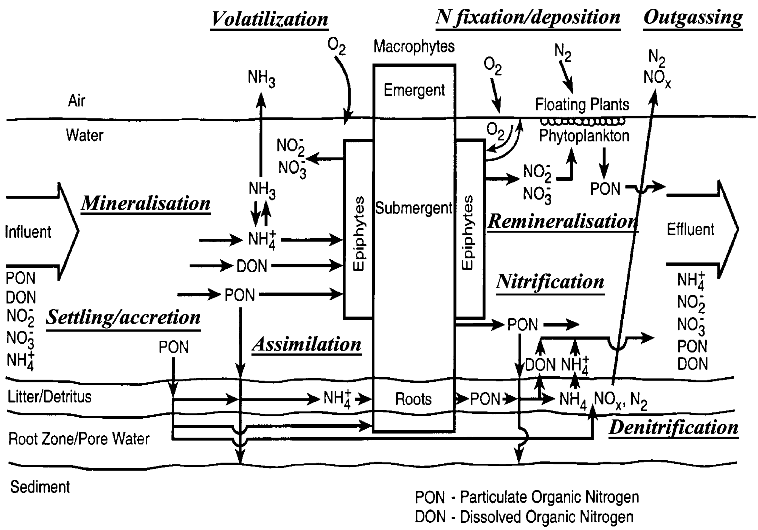

Figure 1: Key nitrogen transformations in SF treatment wetlands (USEPA, 2000). NO_x refers to NO_2 and NO_3-N; PON and DON refer to particulate and dissolved organic forms of N (Org-N).

to N loadings in normal treatment wetlands and the limited measurements of dissimilatory nitrogen reduction to ammonia (DNRA) suggest that it is not likely to be a quantitatively significant process in most treatment wetlands (Bowden, 1986; Tiedje, 1988; van Oostrom & Russell, 1994; Nijburg & Laanbroek, 1997), although it may be under some conditions (Cooke, 1994; Matheson et al., 2002). Reversible adsorption of NH_4-N to sediments, media and biofilms is likely to be a relatively small sink for N under steady state conditions, but may result in rapid removal in systems during start-up and where intermittent loading results in periodic depletion of the sorbed pool (e.g. Tanner et al., 1999; McBride & Tanner, 2000). The dominant transformation processes relevant to constructed wetlands are believed to be:

- *Mineralisation or ammonification* (Org-N $\rightarrow$ NH_4). Anaerobic and aerobic microbial decomposition of organic matter results in the hydrolysis of complex organic forms of N to ammoniacal N.
- *Ammonia volatilization* ($NH_4^+ \rightarrow NH_3$(aqueous) $\rightarrow NH_3$(gas)). As pH climbs above ~8 the proportion of un-ionised ammonia rises rapidly, increasing the potential for volatilization and release to the atmosphere (Jayaweera & Mikkelsen, 1991).
- *Plant and microbial Assimilation* ($NH_4^+ \rightarrow$ Org-N). Nitrogen is an important nutrient for plant growth. It is most commonly taken up by plants in the form of NH_4-N but, as in most terrestrial plants, it can also be taken up as NO_3-N and reduced to NH_4-N internally, or sometimes in organic forms (Marschner, 1995). N uptake and storage by plants can be an important removal mechanism, particularly during the establishment phase, however, unless the plants are periodically harvested and tissues removed, or the N is stored in long-lived tissues (e.g. wood), much of the assimilated N will be returned to the wetland when it senesces and decays. A proportion of the N in decomposing tissues is retained in accreted litter and humic compounds in recalcitrant forms. Assimilation by bacteria and fungi will also occur when there is a surplus supply of organic substrates and nutrients, and also when the microbial pool is expanding. Such immobilisation of N is likely to be minimal once the microbial pool has developed and reached a relatively steady state.
- *Nitrification* (NH_4 ($\rightarrow NH_2OH$) $\rightarrow NO_2 \rightarrow NO_3$). Under aerobic conditions and with an adequate supply of alkalinity, chemoautotrophic nitrifying bacteria can convert ammoniacal N to nitrate (NO_3) via hydroxylamine (NH_2OH) and nitrite (NO_2). Commonly, neither NO_3 nor its intermediaries accumulate in constructed wetlands treating organic wastewaters. This is presumed to be due to the presence of carbon-rich, anoxic conditions that limit nitrification but are highly conducive to denitrification (see below), and to close coupling between nitrification and denitrification at aerobic/anaerobic interfaces.

- *Denitrification* ($NO_3 \rightarrow NO_2 \rightarrow NO \rightarrow N_2O \rightarrow N_2$). This is generally the dominant N removal process in constructed wetlands and involves bacterial conversion of nitrate to N_2O and N_2 gases. Denitrifiers are facultative heterotrophs that use NO_3 and NO_2 as electron accepters in the oxidation of organic matter under anoxic conditions. Although seldom measured directly, available evidence suggests that this pathway may commonly account for 40–90 percent of N removal from constructed wetlands (Tanner et al., 2002). Because it returns N to the vast, relatively inert, atmospheric pool of dinitrogen (N_2) this is generally seen as an ideal, sustainable removal process. However, a proportion of the denitrified (and nitrified) N may be emitted as nitrous oxide (N_2O), which is a potent greenhouse gas in the atmosphere (Houghton et al., 2001).
- *ANAMMOX* ($NH_4 + NO_2 \rightarrow N_2$) *and other alternative pathways*. There is increasing evidence that in oxygen-limited environments nitrification, denitrification and other microbial processes (e.g. methane oxidation) may be much more closely coupled (also described as integrated or simultaneous). They may also include a range of alternative and co-metabolic pathways that overcome oxygen and/or carbon limitations that frequently limit "classical" nitrification and denitrification processes in constructed wetlands.

Anaerobic ammonium oxidation (ANAMMOX) pathways (Fig. 2) have only recently been positively identified in nature (Robertson & Kuenen, 1992; van Loosdrecht & Jetten, 1998; Jetten et al., 1999), despite earlier prediction on thermodynamic grounds. Ammonium oxidising bacteria, which have a higher affinity for oxygen than nitrite oxidisers, are likely to be selectively advantaged under low oxygen conditions. Recent evidence also suggests that "aerobic" ammonium oxidisers have more versatile metabolism than previously assumed, being able to autotrophically denitrify with ammonia as electron donor under oxygen-limited conditions (63% reduction in NOD) or with hydrogen or organic compounds under anoxic conditions, and to use N_2O_4 for ammonium oxidation under both oxic and anoxic conditions (Kuai & Verstraete, 1998; Schmidt et al.,

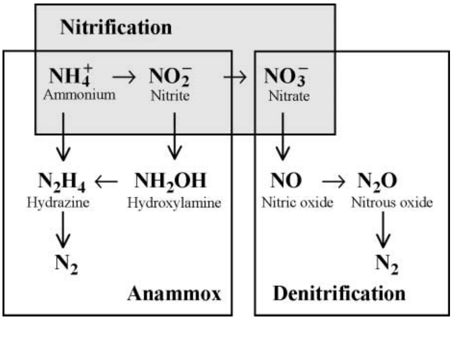

Figure 2: Basic relationship of the Anammox process to nitrification and denitrification.

2002; Sliekers et al., 2002). It has been suggested that "aerobic" nitrifier and anamox bacteria may be natural partners in many oxygen-limited situations (Schmidt et al., 2002), such as those found in many treatment wetlands, and in the root zone of wetland plants generally. Heterotrophic nitrification has been identified, and is sometimes linked directly with denitrification within the same organism (Robertson & Kuenen, 1992). Another option is to "short-cut" the full nitrification–denitrification process and denitrify from nitrite rather than nitrate, thus reducing the oxygen requirement by 25% (Kuai & Verstraete, 1998; Bernet et al., 2001). These alternative pathways need to be investigated further in both natural and constructed wetlands to develop an understanding of their role in wetland N removal.

18.3. N Removal Performance of Constructed Wetlands

General responses of effluent total nitrogen (TN) concentration to N loading are shown in Fig. 3 for North American (USEPA, 1998) and New Zealand (Tanner & Sukias, 2003) SF constructed wetlands treating effluents from domestic and agricultural waste stabilisation ponds. The NZ sewage wetlands for which data was available tended to be relatively highly N loaded compared to those in the North American Wetlands Treatment Database (NADB). Regression equations

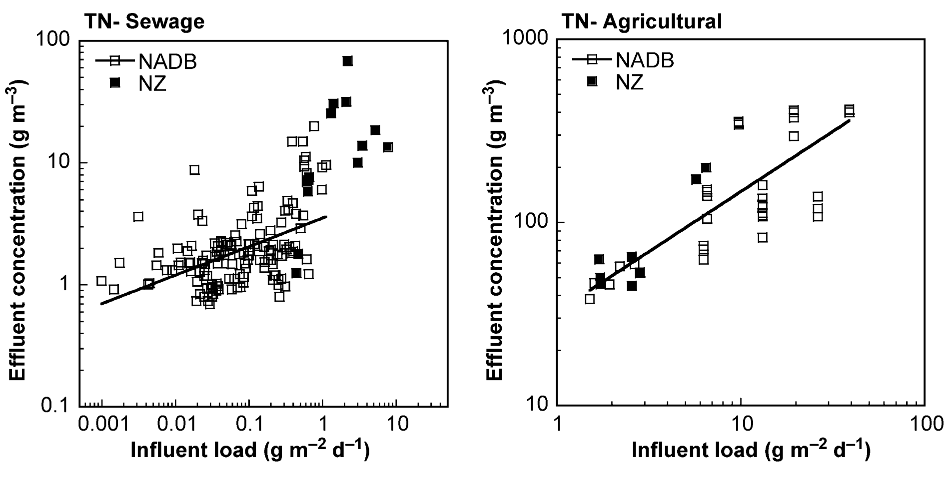

Figure 3: Comparison of mean annual outflow concentration of TN relative to mass loading for SF wetlands treating waste stabilisation pond effluents in NZ and for SF systems reported in the NADB. Each data point is the reported annual average for a specific wetland system or component, and the trend line is a power fit to the NADB data (Tanner and Sukias, 2003).

and rate constants for N removal, derived from North American and European wetland treatment systems are summarised in IWA (2000). These are based mainly on data for systems treating municipal sewage and care should be taken when extrapolating these to other wastewater types where the balance of N forms and/or associated organic loadings (BOD or COD) are different. Considerably higher N removal efficiencies are generally recorded for constructed wetlands treating waste, storm and drainage-waters where N is predominantly present as NO_3-N (van Oostrom & Russell, 1994; Xue et al., 1999; Bachand & Horne, 2000). Ammoniacal N removal can be promoted in wetland systems that incorporate aerobic open-water zones (Hammer & Knight, 1994) or include aerobic phases; e.g. intermittent vertical flow wetlands (IWA, 2000) or fluctuating water levels (Tanner et al., 1999).

18.4. Dominant Mechanisms of N Removal in Constructed Wetlands

18.4.1. Role of plants

TN removal performance for side-by-side studies of planted and unplanted SSF constructed wetlands is compared in Fig. 4 (Tanner, 2001b). Here, despite considerable data scatter, the planted wetlands show a clear trend of improved TN removal.

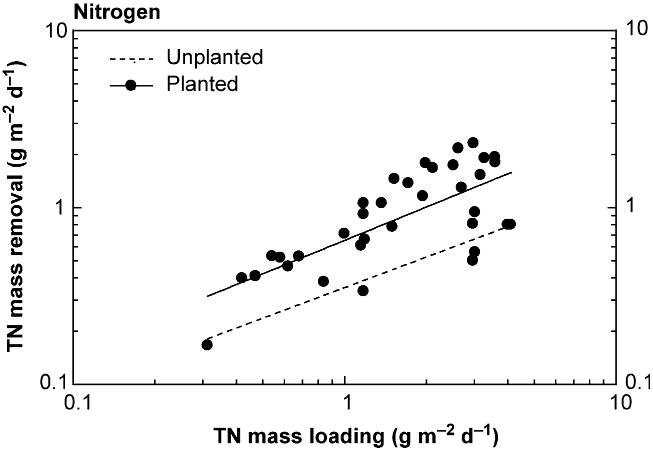

Figure 4: Comparison of mass loading and removal rates of TN for planted and unplanted wetlands. Trend lines shown are power fits (Tanner, 2001b).

The quantity of nutrients that can be taken up and accumulated by live plant biomass per unit of wetland surface area is finite for a given plant species, nutrient regime and set of environmental conditions. Once live plant storage pools approach this limit, little further net annual uptake is possible (Howard-Williams, 1985). In pilot-scale trials where plant storage pools were still actively filling, Gersberg et al. (1986) estimated potential plant uptake could only account for 12–16% of the N removal recorded in SSF wetlands planted with bulrushes. This was 5–7 times less than the additional removal recorded for the planted systems (over that of an unplanted system). In higher loaded SSF systems achieving relatively low N removal, van Oostrom & Cooper (1990) estimated net N uptake by bulrush over an annual period accounted for 25% of wetland TN removal, representing 66% of the additional removal recorded for the planted systems.

Detailed measurements of seasonal uptake by bulrush during the second growth season in four equivalent SSF systems operated over a range of loading rates (Tanner, 2001a) showed that, even in immature systems where plant nutrient pools are actively building, net storage in live plant tissues accounted for only 2–8% of TN removal over an annual period. Net annual plant uptake was responsible for only 3–19% of the additional TN removal recorded for the planted systems. This suggests that plants primarily facilitate improved N removal indirectly via their effects on other removal processes. Plants may enhance N transformation processes (e.g. nitrification and denitrification) through root-zone oxygen release and supply of organic matter. Cycling and accumulation of plant-derived organic matter provides a sustained supply of organic C for microbes (including denitrifiers), sequesters organically bound nutrients, and buffers nutrient release.

18.4.2. Gaseous Emission

Although rarely measured directly in treatment wetlands, microbial denitrification to dinitrogen and nitrous oxide gases is considered to be the primary sustainable nitrogen removal mechanism in wetlands treating wastewaters (IWA, 2000). Ammonia volatilization may also be important in SF wetland systems where the photosynthesis of algae and submerged macrophytes depletes dissolved carbon dioxide in the water column, causing diurnal pH elevation (Jayaweera & Mikkelsen, 1991). In wetlands receiving very high NH_4-N loadings, Poach et al. (2002) found volatilization could account for 12–28% of measured N removal.

Ammonium-Rich Waters. The assumed microbial pathway for NH_4-N removal via denitrification involves initial oxidation to NO_3-N (nitrification). In the predominantly anaerobic waters of treatment wetlands, oxygen availability via atmospheric diffusion and transport through emergent macrophytes is considered to be the main rate-limiting factor for microbial nitrifiers (Gersberg et al., 1986;

IWA, 2000). Because of the presence of abundant organic carbon and reduced compounds, competition for available oxygen is likely to be intense from heterotrophs and other bacteria using alternative electron donors and chemical reductants (Laanbroek, 1990; Adams et al., 1996).

Tanner et al. (2002) attempted to determine the relative importance of different N removal processes along SSF constructed wetlands using experimental cascade mesocosms (wetland tanks in series). Measurements of flow and concentrations of different N species were used, along with a simplified model of sequential N transformations and sinks to infer rates of key N transformation processes down the cascades. When the mesocosms were supplied with four different organic wastewaters, each with contrasting ratios of COD: N and forms of N, it was found that TN and COD mass removal rates varied markedly for the different wastewaters (Fig. 5).

Based on the model, Tanner et al. (2002) found N losses via denitrification accounted for between 60 and 84% of overall TN losses in the cascades, and 0–89% of TN removal in different stages of the cascades. Mean denitrification rates ranged from 0.47–2.0 g N m^{-2} day^{-1} in the different cascades and from 0.0–3.17 g N m^{-2} day^{-1} in individual stages (Fig. 6). Net plant uptake (plant assimilation into above- and below-ground tissues less regeneration from below-ground biomass)

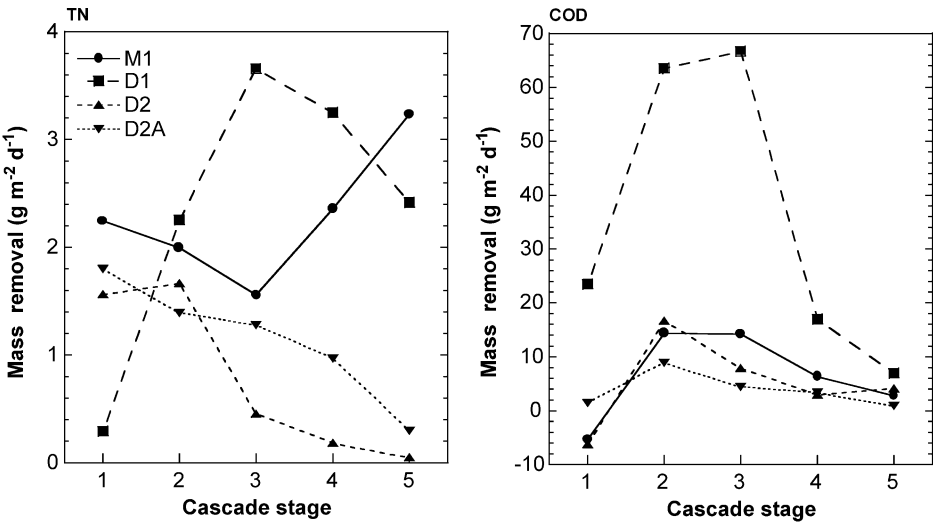

Figure 5: Gradients of TN and COD mass removal during passage through cascade mesocosms simulating horizontal-flow SSF constructed wetlands for four agricultural wastewaters with differing N species balances (Tanner and Kadlec, 2003). The wastewaters had been pretreated in waste stabilisation ponds; M1 = anaerobic-treated meat processing, D1, D2, D2A = anaerobic, facultative and aerated pond treated farm dairy, respectively. See Tanner et al. (2002) for gradients of different N forms.

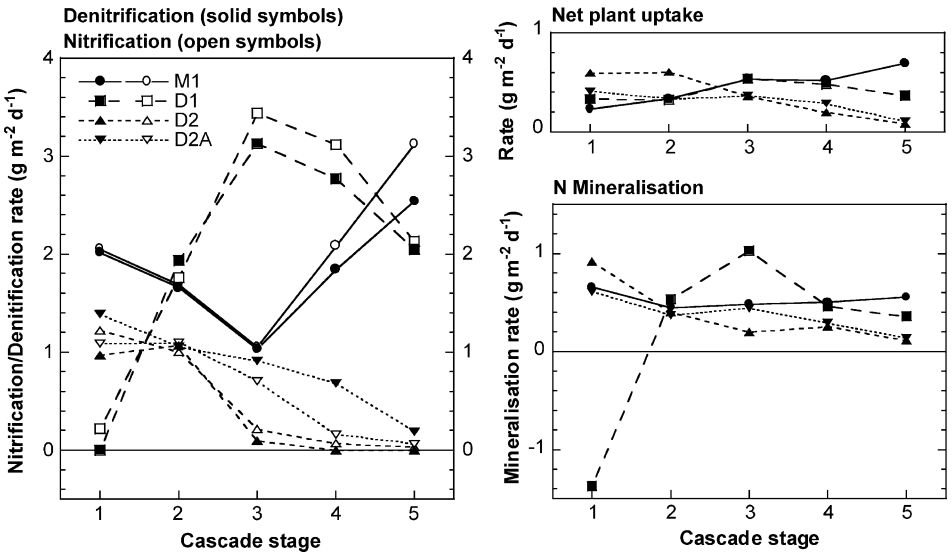

Figure 6: Gradients of key N transformation and removal processes along wetland cascade mesocosms supplied with four different agricultural wastewaters with differing N species balances and simulating horizontal-flow SSF constructed wetlands (Tanner and Kadlec, 2003).

was similar along all the cascades, accounting for $\sim 0.1-0.3$ g N m^{-2} day^{-1} (16–40% of overall cascade TN losses). Mineralisation of organic N along the cascades accounted for $0.1-1$ g N m^{-2} day^{-1}, increasing the realised NH_4-N loading to the wetlands. Apparent negative mineralisation in the D1 cascades occurred, presumably due to net Org-N generation from accumulated organic matter, which was substantial with this wastewater.

In situations where TN mass removal rates were low (less than ~ 1 g N m^{-2} day^{-1}) plant N uptake was an important removal mechanism in the cascades. Apparent N removal via nitrification–denitrification became progressively more important as removal rates increased. This increased the corresponding theoretical NOD required to drive nitrification up to 15 g N m^{-2} day^{-1} in the stages of the cascades where the highest N removal rates were recorded (Tanner & Kadlec, 2003).

Cascades receiving wastewaters with differing characteristics showed contrasting nitrogen process gradients. Overall net plant N uptake, which was likely to have been elevated in the small-scale, harvested mesocosms, represented less than 24% of TN removal in the M1, D1 and D2A, and 40% in D2 cascades. Denitrification accounted for 60–84% of overall TN removal in the cascades, but contrary to commonly accepted paradigms, nitrification was apparently occurring

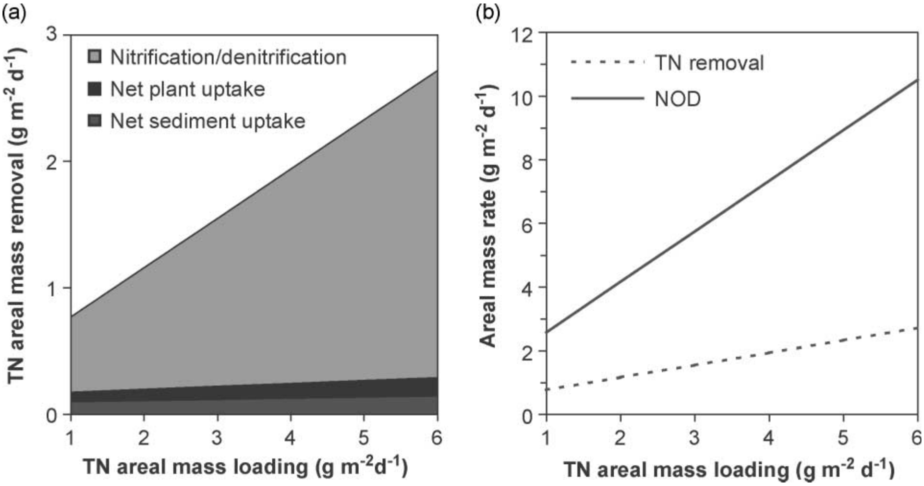

Figure 7: Typical relationships between (a) component TN removal process rates and (b) predicted NOD to wetland TN loading for SSF treatment wetlands (Tanner and Kadlec, 2003).

concurrently with COD removal. General data for N removal in SSF wetlands suggested denitrification is of similar importance in full-scale systems (Fig. 7).

The calculated NOD required to support full nitrification of ammonia and mineralised organic N was in the upper range of that normally able to be supplied by plant root-zone oxygen release. In the organic-rich, predominantly anaerobic environment of SSF wetland beds it is highly unlikely that nitrifiers would be able to compete successfully for more than a small proportion of this oxygen flux (Tanner & Kadlec, 2003). This suggests that oxygen transfer through the wetland surface and via emergent plants is insufficient to support the current paradigm of coupled nitrification–denitrification in SSF treatment wetlands. Better estimates of plant oxygen transport and root-zone release are needed, and the potential importance of recently discovered alternative pathways for nitrogen removal with lower oxygen requirements (e.g. Anammox) need to be explored to improve our understanding of wetland treatment processes.

Nitrate-Rich Waters. Much of the N in mechanically aerated wastewaters, and urban and agricultural drainage waters is commonly converted to the NO_3-N form. Because the process of ammonium oxidation, which is normally rate-limiting, has already occurred, constructed wetlands treating these nitrified waters can generally achieve high removals of N via microbial denitrification (van Oostrom & Russell, 1994; Bachand & Horne, 2000).

Constructed, restored and natural wetlands are increasingly seen as a key tool in the management of diffuse N export from agricultural lands (USEPA, 1993; Mitsch

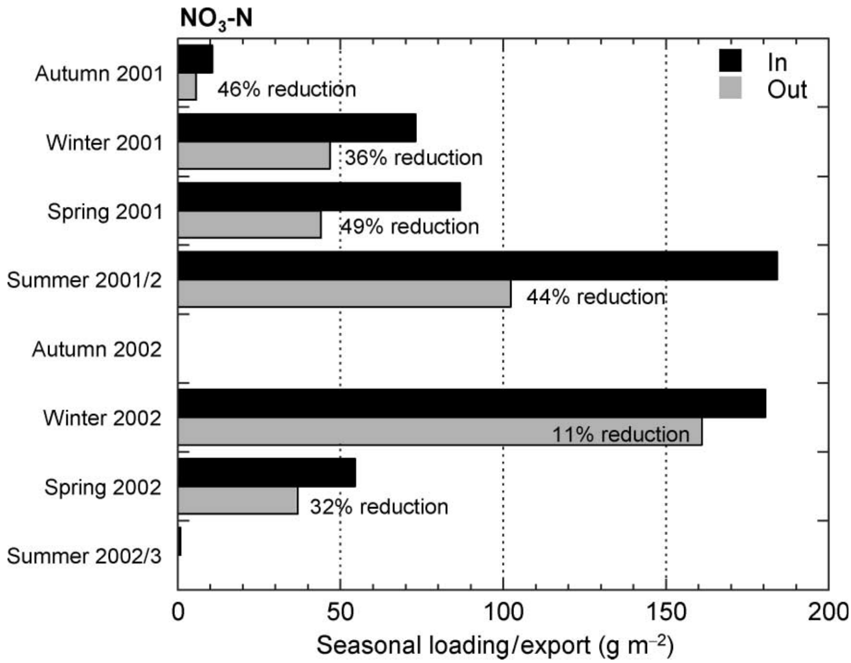

Figure 8: Summary of seasonal mass loads, export and percentage reduction of NO_3-N over 2 years for a constructed wetland treating subsurface agricultural drainage (Tanner et al., 2004).

et al., 2001). Data summarised for a range of North American experimental and field-scale studies (Mitsch et al., 2000) shows NO_3-N removal rates of 95–1,022 g N m^{-2} yr^{-1} for warm climate wetlands and 11–132 g N m^{-2} yr^{-1} for cold climate wetlands. Overall TN removal efficiencies of ~37% TN were reported for an in-stream wetland occupying ~0.8% of an agricultural watershed in North Carolina (Hunt et al., 1999), and also for three wetlands (each ~3% of contributing catchment area) treating cropland tile drainage in Illinois (Kovaic et al., 2000). In a semi-natural wetland (<0.2% of catchment area) treating predominantly subsurface drainage in east–central Illinois, Miller et al. (2002) reported 33% reduction in NO_3-N loads over a 4 year period. Tanner et al. (2004) studying constructed wetland treatment (~1% of catchment) of subsurface drainage from grazed dairy pastures in New Zealand over 2 years found seasonal NO_3-N removal ranging from 11–49%, with overall annual removals of 44% (156 g m^{-2} yr^{-1}) and 16% (52 g m^{-2} yr^{-1}). Variations in the seasonal pattern of N delivery to the wetlands appeared to markedly influence treatment performance (Fig. 8).

Mitsch et al. (2001), summarising data on wetland NO_3^- removal from river waters for multi-year studies carried out in six off-stream wetlands at two sites

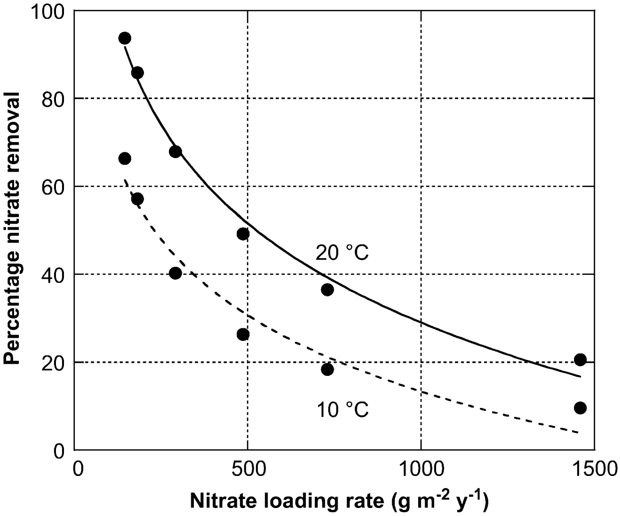

Figure 9: Theoretical relationship between wetland nitrate removal efficiency and loading at 10 and 20°C, based on *k*-C $*$ tanks-in-series kinetic model (IWA, 2000; Kadlec, 2004) with first order areal annual rate constant $K_{20} = 34$, temperature factor $\theta = 1.09$, hydraulic efficiency parameter $N = 3$, influent NO_3-N = 10 g m^{-3}, and areal hydraulic loading rate 40–400 mm day^{-1}.

in mid-western USA, reported NO_3-N removal rising as a power function from ~12–45 g m^{-2} yr^{-1} as loading increased from 20–200 g m^{-2} yr^{-1}. Using data from 65 SF wetlands, including all those noted above, Kadlec (2004) derived a mean first order areal removal rate constant (*k*) of 34 ± 3 m yr^{-1} for NO_3^- removal in surface-flow wetlands, and a mean Arrhenius temperature coefficient of 1.09. Theoretical wetland nitrate removal based on this relationship is summarised in Fig. 9. However the dataset, which included wetlands treating a wide range of NO_3^- concentrations and loadings, water types (wastewaters, stormwaters, agricultural drainage and river water) and flow regimes, showed a wide range of mean *k* values ($<$10 to $>$60 m yr^{-1} for wetlands not receiving carbon supplements). Stormwater flows are characteristically highly variable and pulsed. Further studies are required to better understand constructed wetland treatment responses to such fluctuations in annual, seasonal and day-to-day loads and to develop improved design and performance models.

18.5. Conclusions

- Constructed wetlands can provide effective, low-cost N removal from waste, storm and drainage-waters. N mass removal rates typically rise with increasing

loading up to at least 6 g N m^{-2} day^{-1} (> 2 kg N m^{-2} yr^{-1}) while efficiencies typically decrease from > 80 to < 20% removal.
- In SF wetlands, plants form the main physical structure in the water column, moderating water flow, stabilising sediments, shading and sheltering the water column, and providing surfaces for biofilm growth and organic substrates for denitrifying bacteria.
- In SSF wetlands, plants enhance TN removal rates predominantly through root-zone oxygen release and supply of organic substrates for denitrifying bacteria.
- Direct plant uptake of N is generally of secondary importance for N removal in constructed wetlands, except during initial establishment and prime growth seasons, or at very low N loadings.
- Microbial transformation to gaseous forms is generally the dominant N removal process, except at very low loadings or where elevated pH promotes ammonia volatilization. Estimated oxygen fluxes into SSF wetlands appear to be insufficient to support apparent rates of nitrification (and subsequent denitrification) seen in some studies of wetland wastewater treatment. Emerging information on alternative microbial pathways that operate under low oxygen conditions may help explain these discrepancies.

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Chapter 19

Operation and Maintenance for Constructed Wetlands

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Abstract. Correct operation and maintenance will ensure that constructed wetlands operate as designed and that the objectives are achieved over the life of the system. There are definitive stages in the life of a constructed wetland where management options differ. These include: planning; commissioning; operation; and decommission/retrofitting. This chapter will highlight the importance of operation and maintenance planning. The suggested structure is based on the knowledge gained from the implementation and operation of many wetland systems across Australia. Although management activities required for the operation and maintenance of constructed wetlands within South-East Asia may differ slightly, the structure suggested is relevant for all constructed wetlands, regardless of the location.

Some important management issues of surface wetlands are also addressed including: management of sediment and gross pollutants; plant management; mosquito management; and algal management.

19.1. Introduction

The challenges associated with the successful utilisation of constructed wetlands do not end with design and construction, but go beyond to their subsequent maintenance and operation. Properly managed, constructed wetlands are "passive" and low-maintenance systems. However, they are by no means no-maintenance systems. Constructed wetlands are complex and dynamic with many variables that require managing. Problems will occur when:

- wetlands are poorly designed and constructed;
- the operator has an inadequate understanding of the system;
- the wetland is overloaded, both in terms of hydraulic overload and pollutant overload (White, 1995);

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- natural disasters occur;
- the wetland is plagued by weed problems; and
- excessive amounts of sediment and litter accumulate and are not removed from the system.

19.2. The Phases of Wetland Operation and Maintenance

The operation and maintenance of a wetland can be divided into four distinct phases: planning and construction; commissioning; operation; and decommissioning or major refitting (Beharrell et al., 1996).

19.2.1. Planning and Construction

During the design process many operation and maintenance issues must be considered. Access to all points of the wetland must be provided for operation and maintenance activities. Pretreatment should be provided to formalise areas of sediment deposition and removal of gross pollutants. These must be designed for ease of maintenance. Most importantly water level control must be available.

Poor design, for example, inadequate sizing, will lead to reduced performance and increased levels of operation and maintenance. If the design is inadequate in terms of pollutant removal, overloading will occur and retrofitting or increased pretreatment will be required to maintain the wetland viability.

Currently in Australia many treatment wetlands are constructed by Local Authorities with grant monies from State Authorities. In urban situations wetlands are frequently undersized to fit into the existing environment. A perception exists that something is better than nothing (Seymour, 2000). Generally, these wetlands fail to meet their objectives, have mosquito problems and become maintenance burdens.

A major constraint to surface water wetlands for treatment or habitat can arise from the presence of noxious weeds in vicinity. Weed propagules tend to get carried into wetlands where they can thrive, diminishing visual amenity and wetland function. The presence of certain aggressive weed species may even lead to a constructed wetland being unfeasible.

As with all civil works, attention to detail during construction is required. Failure at this stage will result in on-going maintenance and/or repairs.

19.2.2. Commissioning

This phase will encompass the time from planting to the date at which the wetland is considered operational. During this period, management activities should be tailored to ensure an adequate cover of wetland vegetation. Other specific activities should include:

- "debugging", where minor problems are recognised and resolved, for example structural problems affecting stability of structures in the future — for example rill erosion;
- careful control of water levels to prevent desiccation or inundation of seedlings. Once plants have established, water levels can then be raised to operational levels;
- supplementary planting to fill in gaps; and
- control of weed species which is simpler when infestations are small, surface water wetlands are prone to invasion when the desired vegetation is not well established.

Generally, this period will coincide with plant establishment. Inspections should be frequent and undertaken on a weekly basis, with maintenance being continual.

19.2.3. Operation

The operational phase of the wetland will encompass the active life of the wetland. During this phase the design objectives should be met. Successful maintenance and operation during this phase will prolong the lifespan of the wetland, delaying the need for decommissioning and/or refitting (Beharrell et al., 1998).

19.2.4. Decommissioning or Refitting

At some stage a constructed wetland will either be refitted or decommissioned (White & Kuginis, 1995). For example refitting or decommissioning may be required when accretion of wetland sediments is adversely affecting wetland performance. It is essential that, during this phase, priority be given to public safety. The impact on mosquito production, structural failure and the removal and disposal of possibly toxic sediments should therefore be examined.

19.3. Operation and Maintenance Plans

Operation and maintenance plans are important for the following reasons (Beharrell et al., 2001):

1. they provide direction for the system to be operated as designed and maintenance undertaken to meet the wetland objectives;

2. they save money — early detection of problems will often result in solutions that are much cheaper and simpler to solve than later remedial action. For example controlling weed infestations at an early stage when only sections rather than the entire wetland is affected;
3. changes in personnel can be catered for and loss of corporate knowledge reduced.

An operation and maintenance plan provides the framework for management within which an operator can make decisions based on monitoring, observation, advise and experience. If operation and maintenance of a constructed wetland is to be effective, some party (for example, State or Local authority, private landholder, etc.) who will take on the responsibility for its management must own it.

19.3.1. General Considerations in an Operation and Maintenance Plan

An adaptive approach to the operation and maintenance of constructed wetlands is recommended (White, 1995). Operation and maintenance plans should therefore be flexible and not constitute a full set of instructions or "recipe". Such management plans should be easily understood and considered by the operators as a living document, which can be modified and adapted to the changing needs of the wetland and to information obtained through monitoring and experience (Beharrell et al., 1996).

The operator's understanding/knowledge of the components and processes occurring within the wetland is crucial for informed and effective management. The operator needs to be aware that constructed wetlands do not operate in isolation: its "health" is an expression of the activities occurring within its catchment. The wetland itself can affect the downstream catchment, either through surface water or groundwater, if connected to the wetland.

Not all constructed wetlands share the same goals. They can be designed to satisfy any number of objectives, from water quality improvement, habitat enhancement and/or creation, to aesthetic and educational values. For example wetlands for habitat may require vegetation control to provide roosting and nesting sites for specific wading birds.

The function or objective for which a wetland is designed will determine the kind of management activity undertaken. Management activities will also be influenced by the nature of flow (stormwater, sewage or industrial effluent) entering the wetland (Kuginis, 1998).

Site-specific constraints will affect wetland operation and maintenance. Site-specific constraints include: climate; the presence and/or lack of groundwater; the location; and type (freshwater/estuarine or stormwater) of surface water.

For example, a wetland situated in arid to semi-arid regions may be ephemeral, whilst in other environments, rainfall may exceed evaporation, creating a “perennial” wetland.

There are numerous types of constructed wetlands including surface water wetlands, subsurface wetlands, natural treatment wetlands (Kadlec & Knight, 1996a). These will all have their own particular issues, for example clogging of substrates in subsurface wetlands.

19.3.2. What Does an Operation and Maintenance Plan Need to Contain?

In brief, the essential elements of an operation and maintenance plans are:

- a description of the constructed wetland and its objectives;
- a description of tasks and/or management activities;
- a management calendar indicating when activities are to be undertaken;
- monitoring activities which include maintenance inspection checklists;
- information on overcoming problems that are unique to that particular wetland. For example, control of a particular weed species, ensuring the health and safety of public, etc.

Management Activities. Management activities are those actions, which are required to operate and maintain a constructed wetland, and ensure that a wetland achieves its desired objectives. A management activity sheet can be used to detail the required tasks. It is important that when preparing these activity sheets that the operator outlines for each management activity the following:

- the management zone in which it is to occur (sedimentation, vegetated or open water zone);
- the technique required — what to do;
- the specifications — how to do it;
- the precautions — what not to do.

Management Calendar. Such calendars can be used to timetable works and ensure that maintenance programmes are ongoing, and that staff and/or funds are available. These calendars can be updated and modified as the experience of the operator increases.

Inspection Checklists. Inspections form an important part of monitoring a constructed wetland to ensure continued efficient operation. They should be conducted at regular intervals and after significant events. These include: heavy rainfall; floods; fire; chemical spills and/or events which can adversely impact the wetland. Inspections should be conducted with the aid of a checklist to provide a list of management activities to be undertaken. These checklists provide a permanent record of maintenance activities, which will aid the owner/operator to demonstrate that quality assurance procedures have been undertaken whilst implementing the plan.

19.3.3. Monitoring

Monitoring is required to assess wetland performance, and is therefore an essential part of an operation and maintenance plan. The information collected will need to be interpreted and applied to upgrade operation and maintenance of the wetland (White & Kuginis, 1995).

Monitoring can be undertaken for maintenance, operational control, research activities and compliance with regulatory requirements. The monitoring program(s) will need to be carefully designed so that the most appropriate information is collected (Beharrell et al., 1998). Aspects of the wetland that can be monitored include:

- Wetland components, including operational structures, aquatic plants and pond embankments. These can be monitored for physical changes, e.g. weed invasion, loss of plants and bank erosion. Monitoring can be undertaken through inspections and with the aid of checklists.
- Various performance indicators to aid in wetland operation and to assist in assessing wetland performance. This will define the need for changes in wetland operation and maintenance, and if necessary, the need for refitting the wetland.
- A range of parameters which can be monitored to research aspects of wetland function and performance. In this instance, the design of the monitoring program will be more rigorous to satisfy research procedures where clear research objectives are defined. Research can provide better design criteria for future wetlands and information for wetland modification to improve performance.

19.3.4. Evaluation of Operation and Maintenance Plans

Evaluation of operation and maintenance plans must be carried out after the first year and then every two years. This should be in conjunction with an outside specialist who has expertise in constructed wetlands and will be able advise on ways in which to increase performance and lower maintenance costs. However, for evaluation to be effective the operator will need to document the successes and problems associated with the operation of the wetland, so to provide insight and lessons for improving management.

19.4. Operation of Wetlands

There are relatively few operational variables that can be managed which affect wetland performance. Generally, the only operational variables that can be managed are water level and flow rate.

19.4.1. Water Level Control

Water level management is frequently the only operational variable that can be utilised to influence wetland performance. Water level control can effect residence time, atmospheric oxygen diffusion, plant diversity and plant coverage.

During the summer period or dry season when water levels would naturally be at their lowest, water temperature is usually elevated and plant productivity the highest. Water levels may need to be artificially lowered to encourage regrowth. Reducing water levels can promote better oxygen diffusion to the wetland sediments and plant roots, as the dissolved oxygen levels are low in warmer waters (Kadlec & Knight, 1996b).

19.4.2. Flow Rates

In municipal or industrial waste situations flow rates to the wetland may be regulated. Flow rates can also be controlled by pretreatment, wetland cells in parallel, and recirculation. The regulation of flow rates can affect hydraulic and pollutant loadings. If performance drops or loadings increase flow rates can be reduced to improve performance.

19.5. Management Issues

There are numerous maintenance issues that require management within constructed wetlands. Many of these will be wetland specific and dependent on wetland objectives, design, type of inflows and the catchment characteristics. It must be remembered that there are some situations where inappropriate design and defective construction will require a significant repair or a retrofit.

The operator may be faced with implementing several maintenance activities at once with limited resources. In this case, the activities will need to be prioritised and a decision made on the allocation of resources.

The order of these priorities should be set with reference to the following issues:

- safety — the safety of the public is of the highest priority;
- stability — a failed structure may cause complete failure of the wetland and it is generally cheaper to maintain/repair than to replace;

- plants — deterioration in health or loss of plants increases the risk of not meeting all objectives; and
- all other management activities — essential for the effective long-term performance of the wetland.

Below are the management issues frequently encountered by the author:

- management of sediment;
- litter management;
- vegetation management;
- algal management;
- mosquito management.

19.5.1. Management of Sediment

Pretreatment of inflows is required for the removal of sediment. To ensure that the hydraulic condition of the wetland and vegetation are maintained, removal of built-up sediment will be required. A stormwater wetland with a developing urban catchment will require more regular sediment removal than a wastewater treatment wetland. When removing sediments from a constructed wetland, ensure that sediment is treated as contaminated fill.

Long-term accumulation of heavy metals or unmodified toxic compounds in wetland sediments and vegetation may result in a reduction of these substances downstream. However, the concentrated deposits of toxins may lead to bioaccumulation in the ecosystem of the wetland. Wetland food chains may then redistribute these toxins, endangering fauna and possibly human health (Kuginis, 1998). This may be of particular concern in wetlands for mine drainage.

19.5.2. Litter Management

A gross pollutant trap (GPT) or litter screen will trap significant amounts of the litter entering stormwater wetlands. GPT design must incorporate access for cleaning and maintenance. Periodic cleaning of the GPT will be required as breakdown products from the decomposing litter may re-enter the water column within 10 days of entrapment (Riley, 1995). Consequently, litter removal should be undertaken within 10 days of storm events.

Debris may also accumulate throughout the wetland, especially if there is no GPT. If optimum performance is to be maintained, the litter and debris needs to be removed periodically and immediately after storm events. Fouled areas will have a reduced performance owing to increased hydraulic pressure on the macrophytes and flattening of the plants. Litter removal will also enhance the wildlife habitat and scenic amenity within the wetland environs.

19.5.3. Vegetation Management

Operation and maintenance of a constructed wetland should aim to sustain the presence of desirable wetland plants. Operators should expect gradual changes in wetland vegetation, a result of some species out-competing less aggressive neighbouring species, and recruitment from the catchment. Species diversity may decline in the long term and it is not usually recommended that any action be taken.

In addition constructed wetlands, like natural ones, undergo species composition changes resulting from natural and artificial disturbance. One or a few species may have become dominant: because they arise from disturbed situations constructed wetlands undergo rapid successional changes after establishment.

Water level control, and separation of individual cells can make it possible to maintain plant diversity. Water level control is extremely important, the majority of wetland species will benefit form period of reduced water levels. If water depths are maintained at a constant level the viability of many wetland species will be affected. Logically these should occur in the dry season. However, if plant health is affected, water level reduction may aid in their recovery. Unfortunately, reduced water level can be at odds with the hydraulic design of the system and habitat requirements.

Weed Management. Due to their location in the catchment and the often nutrient rich nature of inflows, surface treatment wetlands and habitat wetlands allow aquatic weeds to flourish (Beharrell, 1999). Within constructed wetlands weeds should be considered as plants that interfere with the objectives of the wetland (Sainty & Beharrell, 1998). For example, mats of floating noxious weeds have the potential to damage structures and affect water quality through the prevention of oxygenation of the water column leading to water quality problems.

The methods used to control weeds will depend upon the species and size of the infestation. The use of herbicides may affect desired vegetation, and manual removal costs may be excessive. Early recognition and control are the most effective measures. Therefore, wetland operators should be familiar with the identification of all species planted in the wetland and aquatic weed species that occur in the general area and/or catchment.

Once present, control of infestations will usually require integrated weed management involving several control methods. Options for control include:

- herbicide application; there are several selective herbicides which can be utilised in wetlands depending on the target species;
- manual removal;
- mechanical removal;
- water level manipulation; and
- biological controls.

Pest Management. There are many potential pest species which can have impact on vegetation within the wetland Aphids and stem borer have been known to attack various plants such as *Phragmite karka* and *Scirpus grossus*, respectively within wetlands in Malaysia. In most cases of insect attack minimum action was taken as predator numbers increased to control the problem. However, if problems persist treatment with appropriate pesticides may be required: this can affect both the desirable biota and the water quality.

Waterfowl are a potential problem at all stages of a wetland's life. Waterfowl damaging plants and defecation of large groups in small areas can affect water quality. In the commissioning phase they can be particularly troublesome decimating seedlings. Using mature plantings may reduce loss to grazing. Alternatively, consider netting the site or avoid the use of sensitive species. Waterfowl do not usually graze some species, such as *Scirpus mucronatas*. In Australia relocation of waterfowl during plant establishment has been used.

Plant Harvesting. Generally, wetland plant harvesting is not undertaken in treatment wetlands as a method of pollutant removal, as it tends to have limited value (Kadlec & Knight, 1996b). Plant take up rates of pollutants is not the major pollutant removal mechanism within wetlands. The disadvantages of macrophyte harvesting include (DLWC, 1995):

- nutrient and sediment re-suspension from disturbance;
- reduction in habitat values; and
- high cost involved.

However, harvesting may be desirable to address other objectives. Thinning of the vegetation and removal of dead material may be required for mosquito control and to improve wetland aesthetics and hydraulics. Harvesting would also occur when a major refit of the wetland is required.

19.5.4. Mosquito Management

Surface water constructed wetlands provide habitat for mosquitoes. Mosquito biology and ecology vary with species, and different species can occupy different niches within a single wetland (Russell, 2000). If there is a perceived mosquito risk, monitoring via larval surveys and strategic population sampling is necessary to determine prevalence and species.

Whilst the initial design plays a major role in reducing mosquito populations, there are some water management techniques that can be utilised to manage populations (Russel & Kuginis, 1998). Techniques include:

- sprinkler or aeration systems that disrupt the water surface;
- water level fluctuations which can be used to control some species, but may increase problems with others (Russell & Kuginis, 1995); and
- periodic draining of the wetlands.

Plant management can also be practised to control mosquitoes. Vegetation can be thinned to allow better access for predators. Floating vegetation and algal mats should be removed which harbour same species. However, as a last resort population can be controlled by chemical and biological larvicides.

19.5.5. Algal Growth

Most surface water wetlands will have blooms of algae at some time because they frequently have ideal conditions for algal growth. These include high nutrient concentrations, shallow water providing idea water temperatures, low velocities, and high hydraulic retention times (Bowling, 1998). Generally, filamentous green algal will be present, occasionally there maybe blooms of Cyanobacteria.

Treatment wetlands frequently have significant algal blooms when first completed. Generally, these subside over time as the system stabilises, and shading from the growing wetland vegetation develops. However, algal growth can have a serious impact on visual amenity in surface water constructed wetlands. Large mats of algae can smother plants and rapidly release accumulated nutrients during decomposition (Sainty & Dalby-Ball, 2000) affecting wetland performance. Decomposing algae can frequently lead to excessive odours.

Generally, the impact on the performance of the system due to the algae is minimal, with treatment only taking place when the wetland objectives are compromised. To minimise algal growth there are several maintenance options available these include:

- Water movement, reducing the water column stability can reduce the incidence of algal growth. Velocity of flows can be increased in systems where this is possible or sprinkler systems can be installed. However, flows should remain relatively low to avoid impacts on wetland physiochemical processes.

- The applications of algicides are frequently used to treat algal problems. These treatments lead to water quality problems and should only be considered as a final measure.
- Shading in shallow areas can be very effective in reducing algal growth. However, the open water areas remain problematic.
- Nutrient removal can impact on algal growth. However, the ability to increase nutrient removal within the system is usually limited. Increase pretreatment of flows maybe considered. There are also several products in the market that can readily reduce the level of nutrients within a wetland in the short-term. For example Phoslock® and Barley Straw.
- Removal of the algae from the open water area can be readily undertaken manually or by machine. This can be incorporated into regular maintenance of the system or as one-off treatments.

19.6. Conclusion

Operation and maintenance activities will change during the lifetime of a constructed wetland. Ultimately the wetland may become inefficient or redundant leading to the site being decommissioned or retrofitted.

To extend the life of constructed wetlands, operation and maintenance must be planned. To facilitate this, an operation and maintenance plan should be developed. This plan must be specific to the wetland and easy to use. In short, the plan should state what is to be done, when it is to be done and how it is to be done.

Monitoring must be undertaken to ensure that wetland objectives are being met and to further define the type and level of maintenance required. Wetland operation and maintenance is more than just cleaning structures. Wetland operation and maintenance relies on attention to detail and an understanding of the functions and processes within the system.

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Chapter 20

Urban and Highway Runoff Treatment by Constructed Wetlands

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Abstract. The pollution removal performance of artificial or constructed wetlands is compared with natural wetland performance in the UK, US and Australia. The reported results for an experimental study of a highway runoff treatment system in the UK show good removal of pollutants during storm events but poor treatment during dry weather conditions. The latter is explained by reference to the relation of inflow pollutant concentrations to the background irreducible concentrations associated with the wetland system. A small-scale experimental wetland study of diesel oil treatment is also described. The design criteria, wetland sizing, optimal hydraulic loading, flow velocity, substrate structure, planting considerations and pre and post-treatment structures in systems incorporating a constructed wetland are discussed.

20.1. Constructed Wetland Types and Flow Systems

Although the design of artificially constructed wetlands varies making each system unique, the basic flow configurations can be divided into two categories:

Surface flow (SF) or *free water surface* (FWS) systems which are similar to natural marshes in that they are basins planted with emergent, submergent and/or floating wetland macrophyte plants. Such free surface water treatment wetlands mimic the hydrologic regime of natural wetlands. Almost all constructed wetlands in the UK for the treatment of urban runoff comprise SF systems, and resemble natural marshes in that they can provide wildlife habitat and aesthetic benefits as well as water treatment. The influent passes as free-surface (overland) flow (and/or at shallow depths) and at low velocities above the supporting substrates. Figure 1 shows a (3×80 m^2) linear SF design which has been retrofitted into a widened stream channel in Dagenham, East London to treat surface runoff from a 440 ha residential and commercial area (Scholes et al., 1999).

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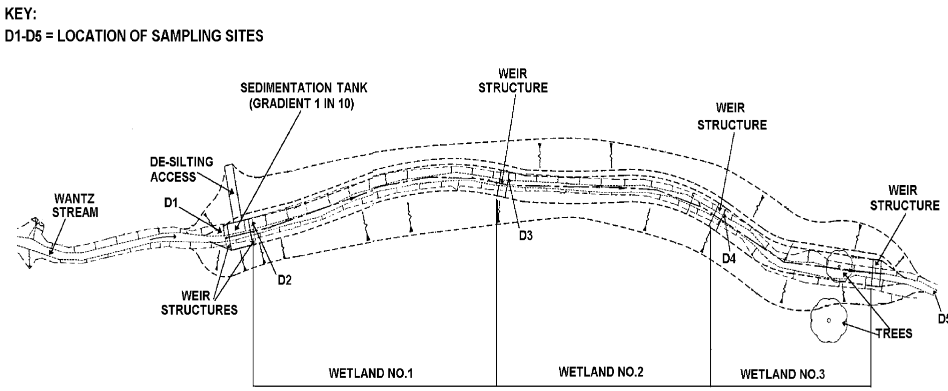

Figure 1: SF constructed wetland design (R Wantz, Dagenham, E London).

The 1,750 m^2 modular wetland system is designed to meet 50% removal efficiencies for targeted pollutants (BOD, Pb, Zn and SS). SF/FWS systems with low flow rates are susceptible to winter ice-cover in temperate climates such as the UK, and have reduced efficiencies during such times since effective water depth and retention time are reduced (Kadlec & Knight, 1996).

Sub-surface flow (SSF) systems operate with the influent flowing below the surface of the soil or gravel substrate. Purification occurs during contact with the plant roots and substrate surfaces, which are water-saturated and can therefore be considered to be oxygen-limited. The substrate in these systems is thermally insulated by the overlying vegetation and litter layer thus the wetland performance is not significantly reduced during the winter. Most of the earliest wetland treatment systems in Europe were SSF systems constructed to treat domestic wastewater. There are two basic flow configurations for SSF wetlands:

Horizontal flow (HF) systems where the effluent is fed in at the inlet but then flows slowly through the porous medium (normally gravel) under the surface of the bed in a more or less horizontal path to the outlet zone. These HF systems are also known in the UK as reedbed treatment systems (RBTS) as the most frequently used plant is the common reed (*Phragmites australis*).

Vertical flow (VF) systems, which usually have a sand cap overlying the graded gravel/rock substrate, and are intermittently dosed from above to flood the surface of the bed. The effluent then drains vertically down through the bed to be collected at the base. Such VF systems are similar in design and operation to conventional percolating filters, but are very rarely found on surface water drainage systems.

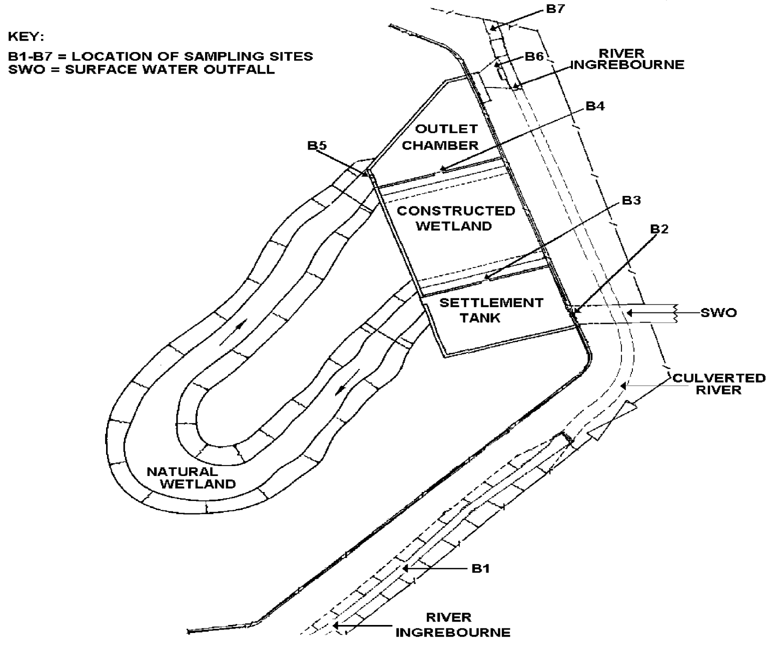

Figure 2: An SSF constructed wetland (Brentwood, Essex).

Fig. 2 illustrates an SSF constructed wetland system located at Brentwood, Essex to treat surface water discharges from a 400 ha mixed urban catchment prior to entry into the River Ingrebourne. During high flows, untreated effluent also overflows into a natural *Typha* wetland in addition to passing through the SSF *Phragmites* wetland before final discharge to the river. The total wetland area is 204 m^2 and the mean retention time is 50 min. Dry weather removals average 30–33% for Pb and Cu, 19% for Zn, 18% for SS, 26% for BOD and 50% for total ammonia with mean metal sediment removals varying between 17 and 33% (Revitt et al., 1999).

20.1.1. Pollutant Removal Efficiency of Constructed Wetland Systems

Table 1 summarises the averages and ranges of removal percentages for various pollutants calculated from the data presented in Nuttall et al. (1997) for constructed wetlands treating domestic wastewater (negative values denote negative efficiencies). The percentage removal efficiency is in most cases simply defined as: $(C_{in} - C_{out})/C_{in} \times 100$, where C_{in} and C_{out} are the inflow and outflow

Table 1: Percentage pollutant removals for domestic wastewater and artificial stormwater wetland systems in the UK.

	SS	BOD	NH_4-N	NO_3-N	*E. coli*
Domestic wastewater					
Secondary treatment	83	82	18	45	68
	(69–94)	(70–92)	(5–29)	(7–68)	(60–75)
Tertiary treatment	68	71	33	55	84
	(25–92)	(50–95)	(0–77)	(40–76)	(46–99)
Urban runoff					
Wetlands	76	24	31	33	–
	(36–95)	(− 57 to 81)	(0–62)	(− 17 to 68)	(52–88)
Combined retention/	73			53	92
detention basins	(13–99)			(10–99)	(86–99)
Wet (retention) ponds	55	40		29	
(with marginal vegetation)	(46–91)	(0–69)		(0–80)	
Extended detention basins[a]					
Highway runoff					
Wetlands (combined	–	18		–	–
retention/detention)	(50–70)	–		(10–20)[b]	(50–90)
SF wetlands	–	15		45[b]	82
SSF Wetlands	(13–75)	(5–32)		(10–60)[b]	(75–99)
	73			53[b]	92
	(13–99)			(10–96)[b]	(86–99)
	85			44[b]	88
	(62–97)			(25–98)[b]	(80–97)

[a] From US data (Urban Drainage & Flood Control District, 1992).
[b] Data for Total Nitrogen.

pollutant concentrations, respectively. The table also shows summary data that have been recorded in the UK for wetland systems receiving urban and highway runoff (Ellis, 1991, 1997; Ellis & Revitt, 1991, 1994; Cutbill, 1994; Cooper et al., 1996). The data for extended detention basins are taken from US data UDFCD (1992) as there are no comparable data recorded for UK sites. The equivalent data for metal removal efficiencies (with ranges shown in brackets and negative values denoting negative efficiencies) that have been noted for various types of surface water wetland systems (Cutbill, 1994; Ellis et al., 1994; Mungur et al., 1995; Ellis, 1999; Heal, 1999; Revitt et al., 1999; Scholes et al., 1999; Halcrow/UPRC, 2000; Revitt & Ellis, 2000) are presented in Table 2.

Although the data exhibit very large ranges, it is clear that artificially constructed wetlands perform better than natural systems, and there is substantial evidence that water and suspended sediment metal concentrations are reduced in urban stormwater wetlands (Shutes et al., 1993; Cutbill, 1994; Hares & Ward, 1999). Some possible concern has been expressed over the ability of urban wetlands to sufficiently remove cadmium, with recorded storm outflow rates frequently exceeding the European Union/Environment Agency for England and Wales water quality standard of 5 μg/l (Revitt et al., 1999; Pontier et al., 2001; Sriyaraj & Shutes, 2001).

A review of a number of studies in the US and Europe suggested that maximum pollutant removal can be achieved in a pre-settlement pond which is equivalent to some 10–15% of the total wetland cell volume (Ellis, 1991). Constructed wetlands in the Environment Agency Midlands Region utilise a stilling pond and sedimentation trap of $10\,m^3$ capacity to capture influent stormwater debris/litter, grit and oiled sediment. This front-end basin can also serve as a back-up spillage containment facility. If sufficient land is available, a final settlement tank (concrete structure) with a minimum capacity of $50\,m^3$ extending across the width of the wetland can be installed. The tank will help prevent fine sediment from the wetland being transferred into the receiving water body. The final settlement tank is an idealised part of the overall system and only needs to be included in the overall design where greatest protection to sensitive receiving waters is required. Regular maintenance is recommended to prevent collected sediments being resuspended during high flows. The rate of sediment deposition will vary with each catchment so the frequency of sediment removal cannot be predicted. Annual inspections should be made to determine if sediment removal is required.

A review of the data reported from international studies broadly confirms the findings arising from the UK wetland database. The results from 26 studies conducted on constructed urban wetland systems in the US have been analysed (Strecker et al., 1992). Although good to high pollutant removal efficiencies were observed, the analysis identified the inherent random nature of the performance data with the absence of any meaningful direct relationships between performance and catchment parameters (Table 3) or with basin/runoff volumes. However, the WWAR and DAR values (see notes below Table 3), are very close to those recommended by European workers who have advocated for example, WWAR ratios of 2–3% and wetland basin volumes (V_b) equal to 4–6 times the mean storm runoff volume (V_r), (Hvitved-Jacobsen, 1990; Ellis, 1999).

Preliminary testing of the US EPA National Stormwater Standardised BMP Database confirms this variability that appears to characterise urban wetland performance (UWRRC & URS, 1999). Table 4, which has been calculated from this 1999 US EPA Database, suggests that this variability is independent

Table 2: Wetland metal removal efficiencies for natural and artificial wetlands in the UK.

	Metals		**Cadmium**	**Lead**	**Zinc**	**Copper**
	Total	**Dissolved**				
Natural wetlands			(−38 to 50)	(−50 to 82)	(−60 to 30)	(10–78)
Artificial wetlands						
Urban runoff						
Wetlands			–	62	57	51
			(5–73)	(6–70)	(−36 to 70)	(10–71)
Combined retention/detention basins			–	–	–	–
			(10–30)	(0–28)	(3–22)	(0–10)
Highway runoff						
Wetlands	–	–	–	69	42	–
	(40–90)	(−15 to 40)	(20–72)	(−41 to 89)	(−36 to 71)	(36–66)
Wet retention basins	–	–		52	38	
	(45–85)	(10–25)		(40–56)	(8–56)	
ED basins	–	–				
	(20–50)	(0–5)				
Dry detention basins (with infiltration)	–	–				
	(70–90)	(10–20)				

Table 3: Reported removal rates for US stormwater constructed wetlands.

	Pollutant removal rates (%)					**WWAR**	**DAR**
	SS	**NH_3**	**TP**	**Pb**	**Zn**		
Median	80.5	44.5	58.0	83.0	42.0	3.65	31.0
CV	27.7	49.4	48.5	56.1	38.8	94.6	156.2
Average	77.1	39.7	57.2	63.8	48.7	4.26	131.0

WWAR, % ratio of wetland surface area to catchment area; DAR, drainage area ratio; CV, coefficient of variation.

of the wetland flow system used although for solids and solids-related pollutants, SSF systems tend to perform better than SF systems.

However, retrofitted "packed bed" SF constructed wetlands in urban flood detention basins in Florida, City of Orlando (1995), have given consistently good pollutant removal rates for SS (78–90%), total nitrogen (63–70%), total phosphorus (62–82%) and total metals (55–73%). Similar horizontal SF wetland retrofitting on 24 sites in the Melbourne urban area of Australia has been successful in reducing pollutant outflow concentrations from detention basins and in improving downstream habitat status whilst maintaining existing flood attenuation capabilities (Wong et al., 1998). Studies in the Sydney region (Shatwell & Cordery, 1999), have indicated average retention in urban SF wetlands of 80 and 60% for SS and Total P, respectively, during small- to medium-sized storm events, but with very variable (and even negative) performance occurring during intense and/or large events.

Table 4: Removal rates for US stormwater SSF and SF wetlands.

	SS (%)	**Total N (%)**	**Total P (%)**	**Faecal coliforms (%)**
SSF systems				
Average	85.4	44.6	50.4	88.5
Range	67–97	25–98	20–97	80–97
SF systems				
Average	73.3	63.3	50.2	92.5
Range	13–99	1.6–99	7–98	86–99

20.2. Experimental Constructed Wetland Studies

20.2.1. Highway Runoff Wetland Treatment Study

The A34 Newbury Bypass in the UK is a 13.5 km porous asphalt surfaced dual carriageway which opened in November 1998. The drainage system includes a series of nine vegetated balancing ponds located adjacent to the highway. Each balancing pond incorporates a front-end oil interceptor and rectangular concrete sediment trap followed by a grassed slope to deliver the highway runoff to the treatment system. A vegetated pond exists as originally designed with a sloping profile which is able to support a variety of fringing macrophytes in the shallows with the predominant species in the main water body (depth: 0.05–1.0 m) being *Phragmites australis*. The original design of a second balancing pond has a constructed wetland which was amended by retrofitting to produce a SSF wetland containing a gravel substrate preceded by a small settlement pond. The constructed wetland was planted with both *Phragmites australis* (front half) and *Typha latifolia* (final half).

Both systems have been assessed by collecting inlet and outlet grab samples during wet and dry weather conditions and automatically controlled storm event samples have been obtained for the constructed wetland (Shutes et al., 2001). Removal efficiencies for suspended solids, Cd, Cr, Cu, Ni, Pb, Zn, nitrate and sulphate for the constructed wetland are shown in Table 5 for the trends observed

Table 5: Comparison of median and dry wet weather removal efficiencies for the constructed wetland.

Parameter	**Median dry weather removal efficiency**	**Median wet weather removal efficiency**
Cd[a]	0.0	84.7
Cr	47.2	42.8
Cu	4.0	−40.3
Ni	72.6	77.5
Pb	0.0	9.1
Zn	5.3	66.2
SS	9.7	57.7
NO_3[a]	5.3	65.5
SO_4	−5.4	44.1

[a] Indicates that the wet removal is significantly better than the dry removal (Mann–Whitney test).

under different weather conditions. The large variabilities in the removal efficiencies derived for both treatment systems, based on the analyses of grab samples, make accurate comparisons of the performances difficult and also raise concerns about using this type of sampling approach for this purpose. Treatment systems are required to function satisfactorily during the increased inlet loadings experienced during storm events, and this is shown to be the case for the constructed wetland for the majority of the monitored pollutants. Despite the existence of performance fluctuations, the generally low levels of inlet concentrations in the highway runoff indicated that the pond discharges did not threaten the environmental quality of the receiving waters.

Chromium and nickel appear to be removed equally well during both types of weather conditions, with Pb showing similar but poor removal performance during dry and wet conditions. In contrast, Cd and nitrate are removed more efficiently during storm events when the data are examined using the Mann–Whitney test, this difference is shown to be significant ($p < 0.05$). There is a similar emphasis on more favourable removal under wet weather conditions for Zn, suspended solids and sulphate although in each of these cases the comparison with dry weather conditions is not significantly different. Only Cu is predicted to have a higher removal during dry weather conditions and this is a consequence of the unexpected behaviour previously described for Cu during storm event monitoring.

The considerations described above assume that the analysis of grab samples obtained simultaneously from inlet and outlet positions during dry weather conditions can be compared directly to storm event monitoring. Ideally, a series of time-based inlet samples should have been collected and compared with similarly obtained outlet samples taking into account the residence time of the constructed wetland under dry conditions. This would have provided a direct comparison between the performances during the two types of extreme weather conditions. In the absence of such a comparison an explanation of the results is not straightforward. Thus, the indicated preferred removal of the two monitored nutrients (nitrate and sulphate) during wet weather would not have been expected as more time for plant uptake would be available during dry conditions and a previous study of the performance of a constructed wetland treating urban runoff has suggested that nitrate removal occurred primarily between, rather than during, storm events (Carleton et al., 2000). Similarly, the settling out of suspended solids should be more efficient under quiescent conditions whereas a higher removal during storm events is predicted by the results. However, this phenomenon is partly a function of the inlet suspended solids concentrations which did not exceed 20 mg/l for routine monitoring but regularly approached 100 mg/l during storm runoff conditions.

Lead is commonly found to be strongly associated with particulate material (Revitt & Morrison, 1987), but the absence of a marked inlet concentration difference between dry (maximum 4.5 μg/l) and wet (maximum 10.1 μg/l) weather conditions results in a median removal efficiency value (9.1%) for the latter conditions which is only slightly higher than the dry weather value (0.0%). Cadmium is the most effectively removed metal during storm events and is most significantly different from the obtained dry weather results ($p < 0.05$; Mann–Whitney test). This finding is again unexpected given the predicted high solubility of Cd in highway and urban runoff (Revitt & Morrison, 1987). Metal removal by a constructed wetland receiving highway runoff can generally be seen to be efficient during carefully designed storm event monitoring conditions (Table 5) with only Cu showing an aberrant behaviour and Pb demonstrating a small positive removal.

The results highlight the limitations of utilising analysed grab samples as the basis for estimating pollutant removal efficiencies between the inlet and outlet of a water treatment system. This is particularly true in wet weather although the automatically controlled sampling of storm events show good pollutant removal in the constructed wetland. There is only a marginal improvement when dry weather conditions prevail both before and during sampling on account of the low inlet concentrations. At such low inflow concentrations, it is difficult to achieve any enhanced removal effectiveness as they represent the minimum or "irreducible pollutant concentrations" (IPC). Such background concentrations (IPC) represent the best performance treatment that can be achieved under low flow conditions and may not be further reduced even if the wetland surface area or volume is increased. Fig. 3 illustrates the range of performance that can be achieved with very variable and even negative efficiencies being associated with inflows (C_0) at or near the background irreducible concentration levels (C^*).

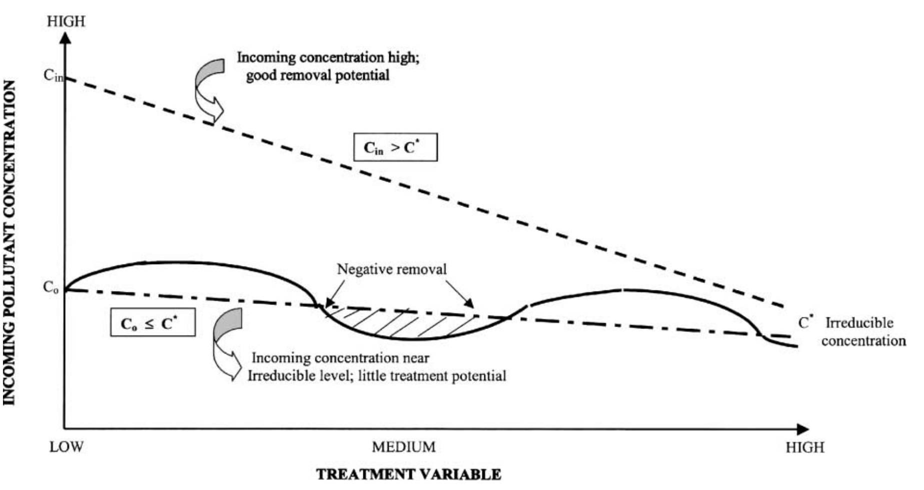

Figure 3: Treatment performance as a function of incoming concentrations.

20.2.2. Small-Scale Experimental Hydrocarbon Treatment Study

The use of constructed wetlands is not yet widely adopted for the treatment of hydrocarbon effluents (Salmon et al., 1998). A monitoring study by Farmer & Roberts (1995) showed 94% removal of oil and grease from cascading ponds colonised by *Typha* spp. and *Scirpus* spp.

The performance of a small-scale constructed wetland for the treatment of oil polluted water was assessed in comparison with an unvegetated system using two outdoor SSF beds (control and experimental, $10 \times 1\ m^2$) filled with a substrate of pea gravel (3–6 mm) to a depth of 60 cm (Omari et al., 2003). The experimental

Table 6: Hydrocarbon removal efficiencies in the top, middle and bottom depths of the experimental and control beds.

	Experimental system			**Control system**		
	Top	**Middle**	**Bottom**	**Top**	**Middle**	**Bottom**
June 1999	60.7	64.2	55.2	57.6	53.2	48.9
Overall	60.0			53.2		
July 1999	72.0	67.5	58.0	63.9	57.3	50.5
Overall	65.8			57.2		
August 1999	91.8	88.8	82.0	88.3	78.0	58.6
Overall	87.5			75.0		
September 1999	85.4	80.2	79.6	84.2	80.5	69.3
Overall	81.7			78.0		
June 2000	78.6	75.5	69.8	71.7	67.7	66.0
Overall	74.6			68.5		
July 2000	83.2	80.2	73.5	78.4	70.9	69.1
Overall	79.0			72.8		
August 2000	89.8	87.2	82.5	81.6	83.4	78.4
Overall	86.5			81.1		
September 2000	74.5	70.7	67.2	71.1	65.9	63.9
Overall	70.8			66.9		
December 2001	85.1	87.9	76.6	54.3	64.7	65.5
Overall	83.2			61.5		
Average overall	80.1 $\pm$9.8	78.0 $\pm$9.1	71.6 $\pm$10.0	72.3 $\pm$11.9	69.1 $\pm$10.3	63.4 $\pm$9.4

bed or small-scale constructed wetland was originally planted with *Typha* seedlings at a density of 7.5 plants/m^2.

Both beds (experimental and control) were treated with the same aqueous concentrations of diesel oil under identical dosing conditions. The average overall hydrocarbon removal efficiencies at the three monitored depths (top, middle and bottom) in the sub-surface systems were 80.1 ± 9.8%, 78.0 ± 9.1% and 71.6 ± 10.0% in the experimental bed, and 72.3 ± 11.9%, 69.1 ± 10.3% and 63.4 ± 9.4% in the control bed (Table 6). The differences in the hydrocarbon removal efficiencies between corresponding months in 1999 and 2000 were statistically analysed and are generally not significant.

The individual hydrocarbon removal efficiencies exceeded 60% in the top sections of both beds except for C-11 and C-25, with C-23 and C-26 also reduced in the control bed (Fig. 4). Overall differences in the removal efficiencies of the planted and the unplanted beds as well as at different depths in both systems, indicate that *Typha*-related removal processes complementing adsorption onto the gravel substrate are occurring.

The results of these two studies of experimental constructed wetlands for highway runoff and diesel oil treatment highlight the need for appropriate and standardised methods of wetland system data collection in terms of sampling equipment, timing and frequency and the location of sampling collection points. Valid comparisons can then be made between the pollutant removal performance of different wetland treatment systems.

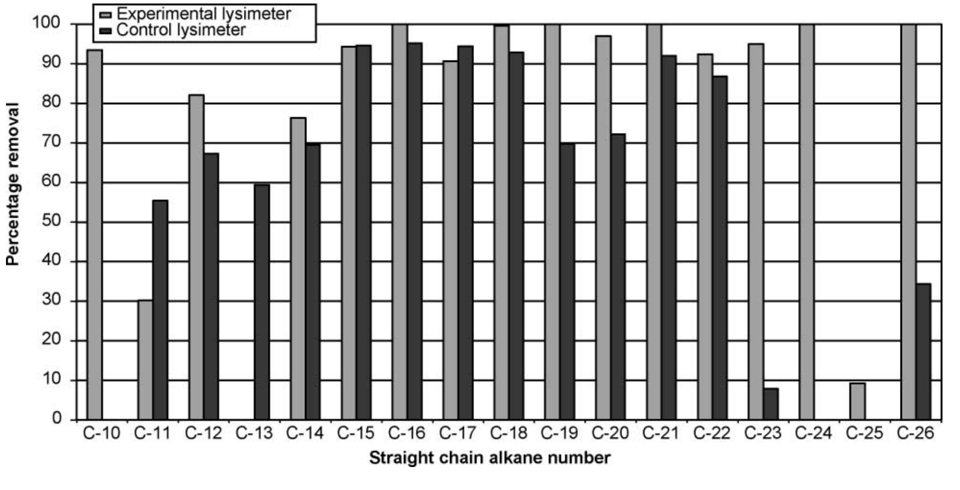

Figure 4: Hydrocarbon removal efficiencies after dosing at the top levels in the experimental lysimeter.

20.3. Urban Wetland Design

The successful design of constructed wetlands for urban surface runoff management requires the adoption of an integrated multi-disciplinary approach as performance criteria are difficult to set given the inherent random fluctuations in discharge and pollution loadings which characterise stormwater runoff. This temporal and spatial variability makes it difficult to define retention time and hydraulic loading, and thus general design rules for urban stormwater wetlands have been developed from empirical performance data and using "single-number" techniques such as drainage area ratio (see Table 3). Thus no UK urban stormwater constructed wetlands are alike in every design respect: a feature readily confirmed from site inspections.

Consideration of water quality issues at the preliminary planning stage can help to mitigate or prevent stormwater management problems in urban catchments and reduce the magnitude and difficulty of surface water treatment. Hydrological effectiveness reflects the competing (and sometimes conflicting) factors of retention time, inflow characteristics and storage volume, and defines the long-term percentage of catchment runoff which enters the wetland basin. Hydraulic efficiency is strongly influenced by basin shape and depth; hydraulic structures such as inlets, outlets and berms; and by the type, extent and distribution of wetland vegetation. Wetland plants are adapted to specific wetting and drying cycles which also significantly influence the organic content and nutrient cycling in the basal sediments. A major factor in determining wetland hydro-cycling (and the overall treatment efficiency) is the interaction between catchment hydrology, basin bathymetry and the hydraulic behaviour (and location) of the outlet structure.

20.3.1. Design Criteria

The most important criterion for the design of a constructed wetland is the selection of the design storm and this in turn determines the wetland size and volume. The objective of the selection process is to determine the critical storm event causing the greatest pollution threat, with this storm event being described in terms of its duration, intensity and frequency of occurrence. In this analysis, it is assumed that the selection process will be based upon single rather than multiple event occurrences. Constructed wetlands can be designed to:

- Retain short duration storms (e.g. less than the 1:1 annual storm event) for the maximum retention time, ensuring that the high flows can be accommodated by the constructed wetland without overland flow in the case of SSF systems or short-circuiting in the case of SF systems. For example, a wetland basin sized to capture 90% of the average annual runoff with a 24-h drawdown would be likely to overflow between 3 and 8 times per year. This would suggest that a feasible design storm for water quality control purposes might be in the order of a 2–4 month storm event.

- Retain longer duration storms ensuring that the initial first flush volume (equivalent to 10–15 mm effective rainfall runoff) containing the heaviest pollution loads receives adequate treatment. It is important that the constructed wetland is large enough to capture the first flush of the larger storm events in order to achieve such partial treatment and to delay outflow discharges to the watercourse, via the wetland and an overflow bypass system, until natural dilution flows have risen.

Where the availability of land and finance is not problematic, the constructed wetland should be designed to treat storms with a return period of 10 years, although the design of attenuation could be up to the 100-year return period. If a compromise is necessary requiring a design based on a shorter return period, the system should be capable of treating the polluted first flush of any storm event. Retention time is an extremely important factor in the treatment performance by constructed wetlands, and even a minimum retention time of only 30 min will help to remove the coarse sediment fractions. Considerations affecting the retention time include the aspect ratio (width: length), the vegetation, substrate porosity and hence hydraulic conductivity, depth of water, and the slope of the bed. Water level and flow control structures, for example flumes and weirs are also required to keep the hydraulic regime within desired parameters. An "ideal" retention time is dependent on the pollutant removal processes operating in the wetland system. Solids sedimentation can be achieved relatively quickly, and a 3–5 h retention will remove a substantial proportion of the coarse solids. However, in order to achieve removal of degradable organics, bacteria and other toxic species associated with the finer solids fractions, much longer retention periods of at least 24 h will be required (Shutes et al., 1997; Halcrow/UPRC, 1998, 2000;). When calculating the retention time in a SSF constructed wetland system, the volume of the bed media must also be taken into account.

20.3.2. Wetland Sizing

The principal problem of wetland design for the treatment of urban and highway runoff is that of optimum sizing given the episodic and random nature of discharge occurrence and the possibility of a rapid succession of inflow events. Sizing is crucial in controlling both the hydraulic loading and retention times needed to give maximum contact and biofiltration/uptake opportunities. The pollutant removal efficiency of an urban stormwater wetland will be directly affected by the frequency, spacing and duration of storm events, all of which are extremely difficult to pre-define. This explains why empirical approaches to the sizing of urban wetlands have been widely adopted. The utility and appeal of such approaches lies in their ability to provide a rapid and robust initial screening methodology for potential wetland alternatives at the early design stages but considerable caution must be exercised in extending them to final design (Kadlec, 2000).

One such approach is to consider the relative percentage of the contributing catchment area or connected impervious area and typically figures of between 1 and 5% have been suggested by Strecker et al. (1992) and Ellis (1999) for this wetland/watershed area ratio (WWAR). Assuming a 2–3% WWAR value, for a 10 ha development site and with retention times equal to 4–6 times the mean storm runoff volume:

$$\text{Surface area} = 100,000\ \text{m}^2 \times 2/100 = 2,000\ \text{m}^2$$

$$\text{Retention volume} = 10 \times 100 = 1,000\ \text{m}^3$$

$$\text{Average wetland depth} = 1,000\ (\text{m}^3)/2,000\ (\text{m}^2) = 0.5\ \text{m}$$

Such sizing criteria would pose considerable land-take difficulties and in any case does not account for any performance considerations.

Nevertheless, it has been shown that such an approach derives hydraulic loading rates (HLR) which are equivalent to the range of HLR values quoted in the national US database (NADB) for point-source SF treatment wetlands (Kadlec & Knight, 1996). They state that as the average annual HLR is close to the mode of the distribution of point-source wetland HLRs, it is reasonable to expect that stormwater wetlands designed using WWAR criteria would perform somewhere near the average quoted for the emergent marsh database set in the NADB. In addition, comparison of the 50 point-source NADB wetland data set with that of 17 urban SF constructed wetlands included in the review by Strecker et al. (1992) for the US Environment Protection Agency, showed very similar efficiency rates when examined on the basis of such empirical design criteria. The mean reduction of total phosphorus in the NADB marsh cells was 57% at an average HLR of 42 mm/day compared to a similar mean reduction of 57% for urban SF constructed stormwater marshes having a 4.3% WWAR value. The equivalent reduction rates for total SS were 81 and 77.1% for the NADB and US EPA wetlands, respectively.

Stormwater wetlands have also been sized to retain water volumes associated with storm events of a specified return period or probability of occurrence. It has been proposed that urban stormwater wetlands should be sized to contain effective runoff up to the 90th percentile value of the design storm event distribution

(Scheuler, 1992). This particular "single-number" design approach has the advantage of allowing a variable percentage of contributing catchment, depending upon the annual rainfall pattern and annual rainfall total. As in the case of the WWAR ratio approach (see above), the derived loading and detention times for SF urban constructed wetlands correspond well with the mean values for point-source treatment wetlands. This is implicitly acknowledged in the listing of pollutant reductions which lie in the mid-range for other types of treatment wetlands, e.g. total phosphorus and total SS removals are quoted as 45 and 69% compared to the 57 and 81% mean cited for the NADB wetland marshes (Scheuler, 1992).

In addition to the design storm and retention time, the following criteria are also recommended:

Aspect ratio (width:length) : 1:4–1:5
Slope of Wetland Bed : 1%
Minimum substrate bed depth : 0.6 m
Hydraulic conductivity of substrate : 10^{-3}–10^{-2} m/sec

Once the design storm and retention time choice has been made, the size of the conceptual constructed wetland can be calculated using Darcy's Law and the above criteria as:

$$\text{Average daily flow rate } (Q_{\mathrm{d}};\ \mathrm{m^3/sec}) = \mathrm{A_c} \times k_{\mathrm{h}}(\partial H/\partial x)$$

where $\mathrm{A_c}$ is the cross-sectional area of the bed, k_{h} is the hydraulic conductivity of the substrate (m/sec) and $(\partial H/\partial x)$ is the slope or hydraulic gradient of the bed (m/m). Darcy's Law assumes laminar uniform and constant flow in the media bed and clean water. In an SF wetland, flow will be channelled and short-circuited and the media will be covered with biological growths, and therefore the equation has only limited usefulness in such wetland design. Nevertheless Darcy's Law does provide a reasonable approximation of flow conditions in SSF constructed wetland beds if moderate sized gravel (e.g. 10 mm pea gravel) is used for the support medium. Fig. 5 provides a schematic section through a SSF constructed wetland illustrating some of these design criteria.

20.3.3. Optimal Hydraulic Loading

During storm events, high rates of stormwater runoff may discharge onto constructed wetlands, but optimal HLR should not exceed 1 $\mathrm{m^3/m^2}$/day in order to achieve a satisfactory treatment (Ellis, 1991). It has been suggested that an arbitrary HLR breakline appears to be about 2.7 ha catchment area/1,000 $\mathrm{m^3}$ storage volume/day, with wetlands having a large area per flow unit (a lower loading

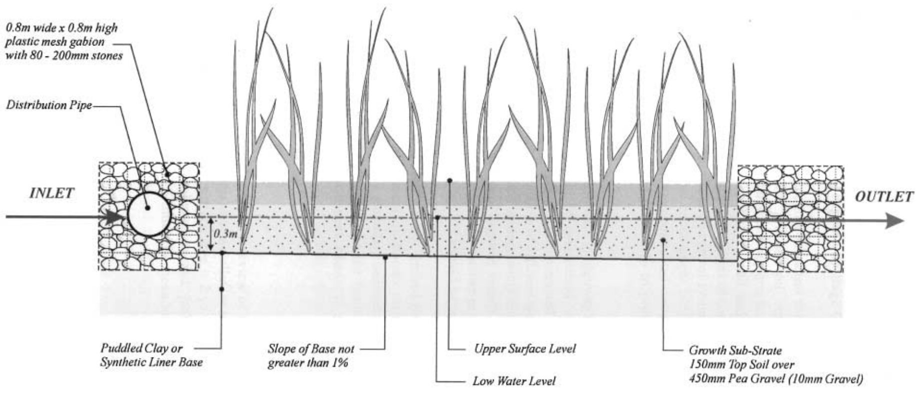

Figure 5: Section through subsurface constructed wetland.

rate) being normally SF systems and smaller areas (with higher loadings) associated with SSF systems (Watson et al., 1989).

20.3.4. Flow Velocity

Flow velocity should not exceed 0.3–0.5 m/sec at the inlet zone if effective sedimentation is to be achieved. At velocities greater than 0.7 m/sec, high flow may damage the plants physically and cause a decline in system efficiency. The inlet pipe should be constructed in such a way that influent flow is evenly distributed across the width of the bed. The level at which the outlet is set is determined by the lowest water level required in the constructed wetland. An additional source of water may be needed to supply the reedbeds during dry periods. Ideally the outlet structure should incorporate control measures which allow the water level in the bed to be varied; a flexible plastic pipe linked to a chain is an appropriate low cost option (Cooper et al., 1996).

An aspect ratio (length:width) of 4:1–5:1 for SSF wetlands and 10:1 or higher for SF wetlands has been recommended for domestic wastewater treatment wetlands. However, any aspect ratio with a good inlet distribution can be applied (IWA, 2000) as previous assumptions that wetlands with high aspect ratios would function more efficiently and be closer to plug flow have not been confirmed from tracer studies. Problems of short-circuiting can be minimised by careful construction, intermediate open-water zones for flow distribution, and the use of baffles and islands.

A grid of slotted plastic pipes (say diameter of 100 mm) should be installed vertically in the substrate (100 mm protruding above the surface, and penetrating the full depth of the substrate) at 5 m centres, to serve as static ventilation tubes and aid aeration of the root zone. Plastic poles should be erected to support lines of bunting to discourage birds from feeding on young plants. The height of the bunting should be about 1.5 m above the substrate surface. Non-metallic items should be incorporated into the construction of the wetland so that metals in the wetland only come from stormwater runoff. Therefore gabions should be encased with geotextiles and the poles supporting bunting should be plastic.

20.3.5. Substrate Structure

Horizontal SF wetlands utilise a natural soil substrate to provide organics and nutrients for plant growth, whereas SSF wetland substrates should primarily provide a good hydraulic conductivity. Nutrient supply can be supplemented to the SSF if required. A combination of organic and clay-based soils, sand, gravels and stones are used in SSF constructed wetlands to provide support for plants, reactive surfaces for complexing of ions and other compounds, and attachment surfaces for microbes which directly or indirectly utilise pollutants. The type of substrate used will have an effect on the hydraulic conductivity and efficiency of the constructed wetland, and must allow for a sufficiently high hydraulic conductivity to enable wastewater to flow at a sufficient rate for treatment without backing up and causing overland flow.

20.3.6. Planting Considerations

Constructed wetlands have traditionally utilised plant species commonly occurring in water bodies and watercourses, which were known to thrive in nutrient-rich situations and were generally pollutant tolerant. The main plant species utilised in sewage wastewater treatment has been the common reed (*Phragmites australis*), which led to the systems being known as RBTS. Reedmace (*Typha latifolia* and *Typha angustifolia*) has been increasingly used, both in sewage-derived wastewater treatment and particularly in the treatment of surface runoff and industrial effluents. Other plant species have played a lesser role in wastewater treatment, such as flag iris (*Iris pseudacorus*), bulrush (*Schoenoplectus* spp.) and sedges (*Carex* spp.).

It is recommended that vegetation for stormwater wetland treatment systems should be selected using the following criteria:

- a rapid and relatively constant growth rate;
- high biomass, root density and depth;
- ease of propagation;
- capacity to absorb or transform pollutants;
- tolerance of eutrophic conditions;

- ease of harvesting and potential of using harvested material;
- growth form (visual appearance);
- ecological value; and
- local retail (or nursery) availability.

20.3.7. Pre- and Post-treatment Structures

Traditional pollution control measures for urban and highway stormwater runoff in the UK have included grit and oil separators for the reduction of sediments and hydrocarbons. They are, however, inefficient in removing the majority of the pollution load and the finer and more mobile sediments and solid-associated pollutants including oil (which clog some designs of constructed wetland treating road runoff). Integrated pollution control systems including a combination of oil separators, silt traps/infiltration trenches, spillage containment facilities and wetland-forebays or lagoons, located prior to the constructed wetland cell(s), can provide for pre-treatment of raw stormwater runoff and help to prevent siltation in wetland inlet zones (Fig. 6) (Halcrow/UPRC, 1998; Shutes et al., 1999; Ellis et al., 2003).

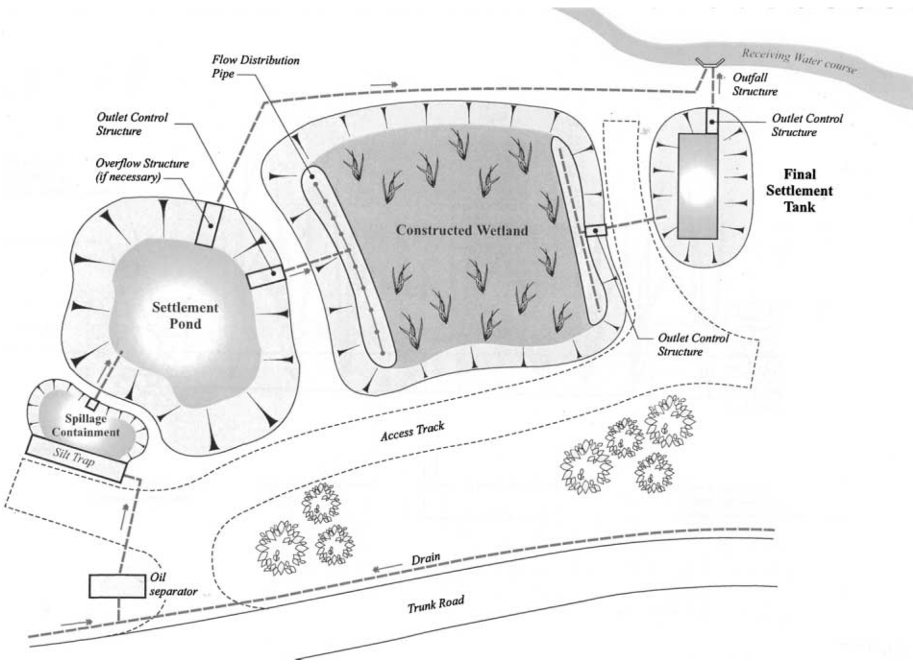

Figure 6: Idealised layout of constructed wetland.

20.4. Conclusion

The current focus on the development of sustainable urban drainage systems (SUDS) in many countries has raised awareness of the advantages of integrating constructed wetlands into urban and highway runoff treatment systems. However, it is essential that the criteria for the selection and design of constructed wetlands are rigorously applied, in order to maximise their pollution treatment performance and maintain and enhance their status as a valuable treatment option.

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Chapter 21

Wetland Ecosystems for Treatment of Stormwater in an Urban Environment

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Abstract. The drive towards urban consolidation has placed increased emphasis on development of innovative stormwater management solutions. This chapter describes management strategies undertaken by the NSW Department of Public Works & Services (DPWS) in utilising wetland ecosystems integrated into the general landscaping/streetscape for the treatment of urban stormwater. The strategies are based on formulating designs that incorporate multiple objectives and opportunities for integrating the following: urban design, landscape, aesthetics, engineering of subsurface ground conditions and stormwater runoff into the overall management strategy, so that a total water cycle management approach can be adopted, for an existing or new sub-division within an urban environment. It should be noted that the management strategies could also be combined to form a Universal Stormwater Treatment Train Model. The chapter also introduces the Victoria Park project in Sydney, Australia, which was constructed with these wetland ecosystems for site stormwater management.

21.1. Introduction

It is widely recognised that urban environments generate increased stormwater runoff and contaminants/pollutants, which can cause negative impacts on the aquatic ecosystem of receiving waters (Gan, 1998). The management of urban stormwater for water quantity and quality control/improvement, are now becoming standard considerations in urban design. This chapter describes the management strategies undertaken by the NSW Department of Public Works & Services (DPWS) in utilising wetland ecosystems integrated into the general landscaping/streetscape design for the control/treatment of urban stormwater (Gan, 2001a). The chapter also introduces the Victoria Park project in Sydney, Australia, which was constructed with wetland ecosystems for site stormwater management.

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21.2. Stormwater Pollution

A number of pollutants are typically found in urban stormwater runoff. These pollutants originate from either point or non-point sources. Point sources are specific identifiable locations where stormwater pollution can occur, e.g. include illegal discharges of trade wastes and sewer overflows. Non-point sources or diffuse sources, are more general, and are comparatively difficult to identify and control, e.g. include litter, sediments, nutrients, oils and grease from road surfaces, toxic material, bacteria and organic material. Without appropriate stormwater treatment devices, the resulting impacts on receiving waters can be devastating, not only for aquatic ecosystems but also to community values such as aesthetics, recreation, economics and health of receiving water bodies.

Stormwater pollution can come from a variety of sources. Table 1 provides a summary of some of the land use activities and the likely pollutants that may have negative impacts on the environment. This information can be used to establish targeted strategies to reduce the pollutant loads on the environment, e.g. based on the catchment characteristics and the contaminants, so that a suite of treatment strategies can then be tailored to control/trap the specific contaminant.

Table 1: Relation of land-use to contaminants/pollutants generated (Gan, 1999).

Land uses	**Contaminant/Pollutants**						
	Litter	**Sediment**	**Nutrients**	**Oil and grease**	**Toxic material**	**Bacteria**	**Organic material**
Residential	x	x	x	x		x	x
Industrial/ Commercial	x	x		x	x	x	
Open space	x		x			x	x
Roads	x	x		x	x		
Raw sewer overflows			x	x	x	x	x
Construction activities	x	x		x			
Land fill	x	x	x		x	x	x
Septic tanks			x	x	x	x	x
Underground storage tanks				x	x		

Note: “x” denotes likely contaminants produced from the specified land use activities.

21.3. Management Options

The identification of stormwater pollutants, and likely source/s, will enable the selection of appropriate strategies for managing or trapping the pollutants generated from urban stormwater, and thus protect the water quality of the receiving waters.

A variety of options are available for addressing the stormwater quality management issues (Gan, 2001a). The approach to water quality improvement adopted by DPWS has been undertaken in two separate approaches. The *first approach* looks at ways in which pollutants entering the stormwater system can be reduced, and the *second approach* looks at the pollutants once they have entered the stormwater system, and considers how can they be removed before they enter receiving waters. Clearly "prevention is better than cure" — not only is it better, i.e. more effective, it is also cheaper. The cost for removal of pollutants trapped is substantial and it is not only a one off capital cost — there are ongoing associated maintenance/cleaning costs that can be substantial. The options are:

- *Non-structural*. Potential non-structural options include:
  - educational measures (e.g. advertising in local papers, radio and TV media, school curriculum, etc.);
  - planning controls (e.g. council policies and strategies etc.);
  - site auditing;
  - review of management practices (e.g. council maintenance/cleaning activities, etc.);
  - studies and assessments;
  - others.
- *Structural*. Structural options (NSW DPWS, 2002) for stormwater management can be beneficial for targeting known "hotspot" locations within the catchment. These solutions typically address the immediate, and often visible, issues as opposed to addressing the source of the problem. Some structural options include:
  - at source controls:
    - litter traps, e.g. litter basket, litter booms, nets, trashracks;
    - pit inserts — from Enviropod, Dencal industries, Ecosol, Net Tech, etc.;
    - bank stabilisation, e.g. vegetation planting, gabions and reno mattress, etc.;
    - silt fences and sand filters;
    - buffer strips, grass swales, bio-retention/infiltration, wetlands;
    - universal stormwater treatment trains.

- At "in-line" or "end-of-pipe" controls:
  - gross pollution traps,
  - booms,
  - sediment traps,
  - constructed wetlands, and
  - universal stormwater treatment trains (i.e. a combination of the above).

21.4. Selection of Management Options

The first step in selection of a management option, is to decide what the target pollutants are. A wide variety of pollutants have been identified, as being washed off from urban catchment/s by the action of rainfall and stormwater runoff. Stormwater pollutants typically found in urban catchments include those listed in Table 1.

At present, it would be uneconomical to select a stormwater treatment device to capture all the stormwater pollutants listed in Table 1. So, the normal practice is to target only the gross pollutants. Gross pollutants can be defined as all the substances listed in Table 2. The table also lists the criteria and trapping requirements for gross pollutants (NSW EPA, 1996, 1998).

Table 2: Gross pollutants and capture requirements.

Gross pollutant	Description	Capture and trapping criteria
Litter	All anthropogenic material, e.g. cans, bottles, plastic bags, etc.	Capture 100% of average annual litter load greater than 5 mm
Coarse sediment	Coarse sand (particles between 5 and 0.5 mm)	Capture 90% of average annual load for particles 5–0.5 mm
Medium sediment	Medium-sized soil (particles between 0.5 and 0.062 mm)	Capture 75% of average annual load for particles 0.5–0.062 mm
Fine particles	Fine sand (particles smaller than 0.062 mm)	Capture 50% of average annual load for particles 0.062 mm or less
Nutrients	Total phosphorus and total nitrogen	Capture and retain 45% of average annual load
Cooking oil and grease	Free floating oils that do not emulsify in aqueous solutions	Capture 90% of average annual pollutant load with no visible discharge
Hydrocarbons, motor oils and grease	Antropogenic hydrocarbons that can emulsify	Capture 90% of average annual pollutant load

21.5. Issues and Causes

Table 3 provides a summary of the issues, potential negative impacts and possible causes on the environment based on the ecological, social and administrative concerns.

21.6. Design Terminology

Treatable flows are used and normally associated with flow hydraulics and the pollution capture effectiveness of a particular stormwater treatment device, i.e. if the stormwater treatment device is able to allow more treatable flows through the structure, the higher the level of pollutant removed and captured. The minimum treatable flow rate for a stormwater treatment device should be quoted for a three-month average recurrence interval (ARI) storm event or tied-in with the maintenance programme. However, in DPWS Contract 019, there has been an array of flows quoted by the different manufacturers. These flows should be re-defined, e.g. suggested definitions include:

- design flow — similar to as defined for treatable flow rate, i.e. design should be for a minimum of a three-month ARI storm event or tied-in with the cleaning programme;
- maximum flow — maximum flow that can pass through the stormwater treatment device, safety without causing any major damage to the structure, i.e. 100-year ARI storm event;
- hydraulic capacity — similar to as defined for maximum flow, i.e. that a 100-year ARI storm event can pass through the stormwater treatment device, safety without causing any major damage to the structure.

21.7. Pollutant Loading Rates

Table 4 shows the pollutant loading rates obtained from various sources including our on-going research and development.

21.8. Wetland Ecosystems for Treatment of Stormwater

As discussed, this chapter describes the management strategies undertaken by DPWS in utilising wetland ecosystems integrated into the general landscaping/streetscape design for the treatment of urban pollutants, e.g. buffer strip, grass swale, bio-retention/infiltration, wetland and the universal stormwater treatment train approach.

Table 3: Summary of the issues, potential negative impacts and possible causes (DPWS and DLWC, 1997).

Issue	Potential negative impact	Possible cause
Litter and debris	Reduces aesthetic appeal of waterways Can kill some marine aquatic life (e.g. fish, turtles, sea birds) Decay of some gross pollutants can decrease dissolved oxygen levels	Littering, e.g. bottles, plastic wrapping and caps, cigarette butts Overflowing rubbish bins Waste dumping Uncovered loads (e.g. trucks, trailers)
Sediment deposited in the bottom of receiving waters	Smothering of plants and animals that live on the bottom of receiving waters, ponds, lakes and streams	Erosion of sediment from building sites Erosion from bare earth areas, e.g. unsealed roads, driveways and car parks, poorly maintained lawns
Turbidity in waterways	Reduced aesthetic value (water looks "muddy") Reduced aquatic plant growth Clogging of fish gills Hinders the ability of aquatic predators (e.g. certain fish species) to see their prey	Soil and sand piled on nature strips, footpaths, driveways and gutters Washing cars in the street Air pollution carried by rain into stormwater systems
Nutrient enrichment	Nitrogen and phosphorus stimulates the growth of algae and aquatic plants Decay of algae and plant matter reduces dissolved oxygen levels Excessive growth of algae and aquatic plants reduces waterway aesthetic values	Washing cars with detergent containing phosphorus. Excessive use of fertilisers, which is washed off lawns Decay of plant material Leaky or overflowing sewerage systems
Petrol, oils and grease	Reduces aesthetic appeal of waterways Can harm some aquatic life Decay of some hydrocarbons can decrease dissolved oxygen levels	Leaks from vehicles Car washing or maintenance Illegal dumping of waste; lubricating or food oils
Pesticides and herbicides	Harms aquatic plants and animals	Pesticides and herbicides (weed killers) used on gardens and nature strips and washed off during rain

(*continued*)

Table 3: Continued.

Issue	**Potential negative impact**	**Possible cause**
Trace metal pollution (heavy metals)	Stress on aquatic plants and animals Contamination of the food chain with trace metals	Runoff from roadways or car parks Deterioration of building surfaces (e.g. rusting galvanised iron roofs) Byproduct of burning fossil fuels Swimming pool water
Bacteria and other pathogens	Makes contact with water unsafe for humans Causes disease in aquatic organisms Contaminates shellfish	Animal (dog and cat) faeces Food wastes disposed improperly Leaky or overflowing sewerage systems
Vegetation washed into waterways	Oxygen dissolved in the water is used up when plant matter decays. Fish and other water life need this oxygen to live	Leaf drop from gardens and street trees, particularly when they fall onto paved surfaces Hosing or sweeping lawn clipping and leaves into gutters Mulch washed or blown from gardens
Loss of aquatic habitats and/or riparian vegetation	Weed infestation of urban bushland Nutrients in stormwater and the transport of weeds propagated from urban areas by stormwater Reduced fish population Water pollution due to loss of filter; strips adjacent to creeks Increased water temperature	Riparian vegetation cleared Changed flow characteristics resulting from change of land use Bank erosion Unrestricted access Creeks "channelised" or "piped"
High runoff rates	Increased pollutant loads Erosion of creek banks roads Changed pattern of water levels in wetlands, affecting aquatic flora and fauna Increased frequency of disturbance to aquatic ecosystems, reducing the diversity of aquatic life	Increased impervious surfaces (e.g. roofs, paved areas, footpaths) directly connected to the stormwater system

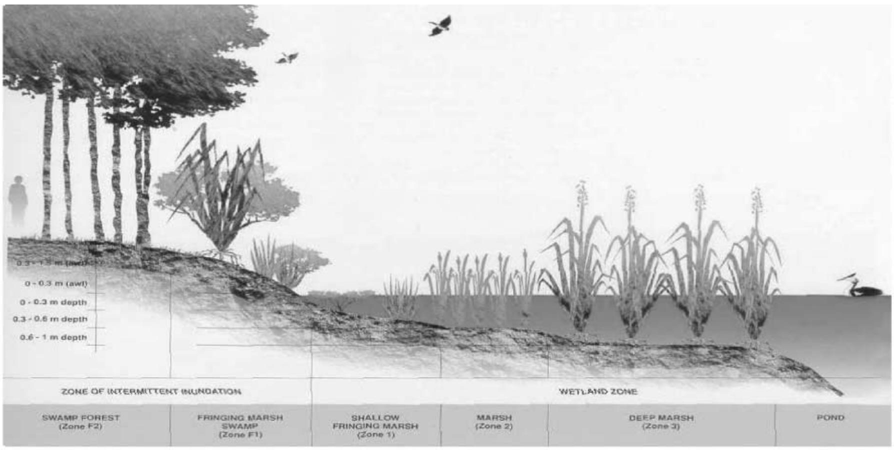
Table 4: Pollutant loading rates (kg/ha/yr).

Pollutant	USEPA (1983)	CSIRO (1991)	Water Board (1992)	Willing & Partners (1993)	CRC (1996)	EPA (1997)	Sydney Water (1998)	DPWS (2002)
Sediment					750		230	1200
Suspended solids	1731	120	200	500–600	200	500		500
BOD	72.3		25	32–40	40			30
TN	14.63	9.3	9.3	9.5–12	8.5	12		10
TP	2.85	1.2	1.2	0.76–0.97	0.85	1.65		3
Lead	0.424		0.4					0.5
FC (cfu/ha/yr)*	0.41×10^{12}*		3×10^{12}*	0.22×10^{12}*	0.75×10^{12}*	2.15×10^{12}*		0.5×10^{12}*
Gross solids or litter (m^3/ha/yr)*							0.11*	0.33*
Organic material (m^3/ha/yr)*							0.32*	2.04*
Hydrocarbons								5

*Denotes pollutant loading rates as stated.

21.8.1. Buffer Strip

Fig. 1 shows the buffer strip. These are vegetated areas that treat overland (sheet) flow, and commonly used as a source control measure, particularly adjacent to water courses or management of road runoff. They are effective in the removal of coarse to medium-sized sediments and can be used as an effective pre-treatment measure.

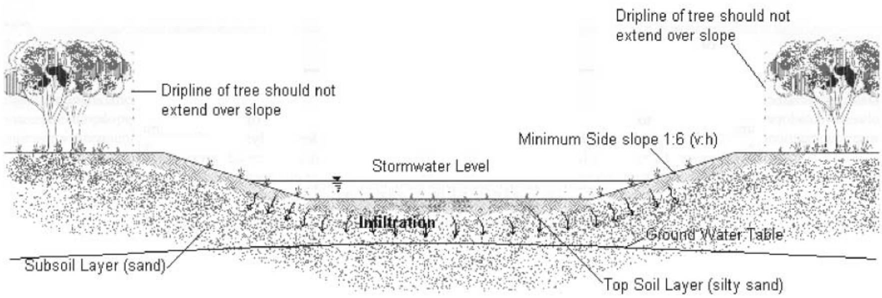


Pollutant	*Trapping efficiency*	*Pollutant*	*Trapping efficiency*
Litter	L-M	Oil & grease	H
Oxygen demanding material	L	Nutrients	M
Sediment	H	Bacteria	H

Figure 1: Buffer strips.

21.8.2. Grass Swale

Fig. 2 shows the grass swales. These are vegetated open channel systems, which utilise the grass to aid the removal of sediment and suspended solids. These systems are a subjected to fairly high hydraulic loading and the removal efficiency is highly dependent on the density and height of the grass.

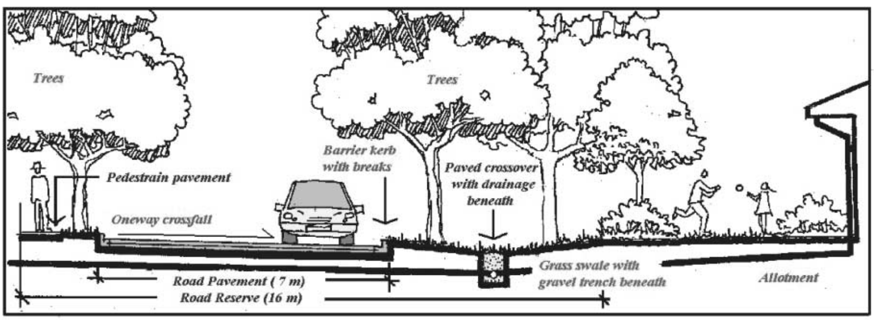


Pollutant	*Trapping efficiency*	*Pollutant*	*Trapping efficiency*
Litter	M	Oil & grease	H
Oxygen demanding material	L	Nutrients	M
Sediment	H	Bacteria	H

Figure 2: Grass swale.

21.8.3. Bio-retention/Infiltration System

Fig. 3 shows the bio-retention/infiltration systems. These systems promote the removal of particulate and soluble contaminants by passing stormwater through vegetation and filter medium. The type of vegetation and filter medium determines the effectiveness of pollutant removal, with the vegetation and material of lower hydraulic conductivity providing the most efficient pollutant removal (owing to longer detention time). Typical filter material ranges from gravel ($\sim$10 mm) to fine sand ($\sim$0.1 mm).

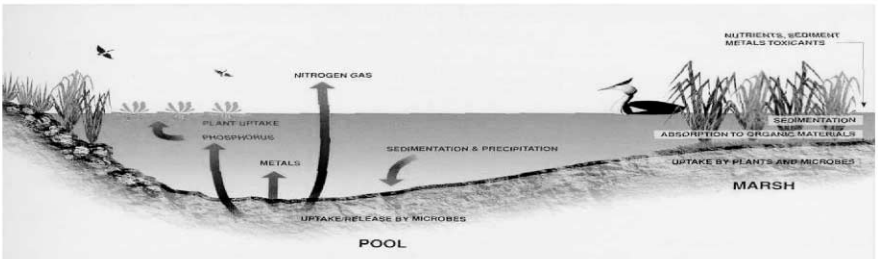


Pollutant	*Trapping efficiency*	*Pollutant*	*Trapping efficiency*
Litter	N-L	Oil & grease	H
Oxygen demanding material	H	Nutrients	M
Sediment	H	Bacteria	H

Figure 3: Bio-retention/infiltration system.

21.8.4. Wetland

Fig. 4 shows the wetlands. These are an effective stormwater treatment measure for the removal of fine suspended solids and associated contaminants, as well as soluble contaminants. These systems utilise a combination of physical, chemical and biological processes in removing stormwater pollutants. They are used as “end-of-pipe” or at “source control measures” (DLWC, 1998).



Pollutant	*Trapping efficiency*	*Pollutant*	*Trapping efficiency*
Litter	L	Oil & grease	H
Oxygen demanding material	M-H	Nutrients	M
Sediment	H	Bacteria	H

Figure 4: Wetland.

21.8.5. Universal Stormwater Treatment Train

These can be defined as the integration of best management practices (BMP) to achieve management objectives. The objectives may include:

- water quantity control — avoidance of flooding;
- water quality improvement — all water discharges to have no impact on receiving waters;
- conservation — optimise the use of rain that falls (i.e. apply reuse strategies);
- protection and enhancement of natural water systems — preserve natural drainage eco-systems;
- improving aesthetic, incorporating social and ecological objectives — provide an opportunity for the community to gain an enhanced appreciation of water as essential element of the urban environment.

Typical examples of BMP are as follows:

- for water quantity control — grass swales, adsorption pits, bio-retention/infiltration systems, detention and retention basins;

- for water quality improvement — gross pollutant traps, bio-retention/infiltration systems, oil and grit separators, water pollution control ponds, sediment traps, wetlands;
- for conservation — rainwater tanks, water re-use;
- for protection and enhancement of natural water systems — trashracks, bio-retention/infiltration systems, constructed wetlands;
- for improving aesthetic, incorporating social and ecological objectives — grass swales, bio-retention/infiltration systems, constructed wetlands.

21.9. Victoria Park Project in Sydney, Australia

21.9.1. Project Description

The Victoria Park site occupies an area of 24 ha. The site drains into the Shea's Creek–Victoria Branch drain just beyond the south-western corner of the site in Joynton Avenue. Flooding has been reported at the low point along South Dowling Street near the Winkurra Street intersection and at the low point along Joynton Avenue south of Elizabeth Street. Flooding appears to be due to the limited capacity of the Victoria Branch drain, which crosses Joynton Avenue at the low point, the limited inlet capacity of the street entry structures and the high ground water table. Fig. 5 shows the proposed re-development and the site drainage conveyance/treatment system.

The proposed drainage infrastructure has been designed to cater for all flows, i.e. the off-site and on-site stormwater runoff flows into and out of the development site, designed to cater for the 100-year ARI storm event and also to treat the stormwater pollution to meet ANZECC standards (ANZECC, 1992). In general the drainage infrastructure will consist of the following elements:

- a series of trashracks to trap rubbish and litter;
- gross pollutant traps to trap litter, grease, oils and coarse sediments;
- a series of buffer strips and grass swales;
- a series of bio-retention/infiltration system that provide drainage conveyance and stormwater;
- a piped drainage conveyance system to cater for major flows up to the 100-year ARI;
- an 8,000 m^3 storage onsite detention basin to cater for all flows, i.e. the off-site and on-site stormwater flows;
- water control appurtenant structures consisting of pits, grates, valves, etc;
- a water recycling system consisting of pumps and mains for the on-site water reuse and irrigation system.

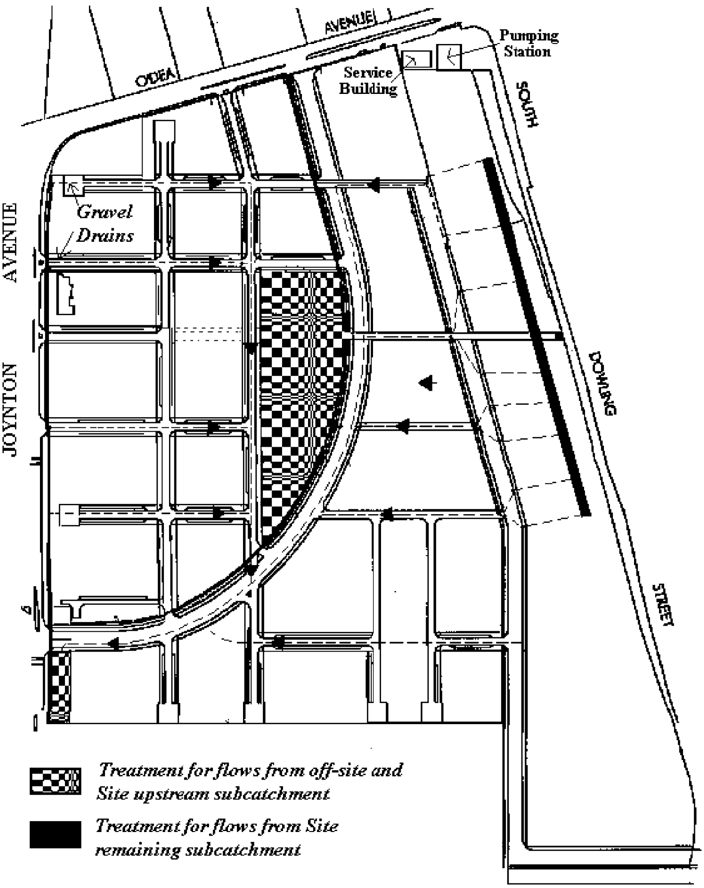


Figure 5: Site drainage conveyance and treatment system.

21.9.2. *Purpose*

The objective is to construct an overall drainage system that will integrate the planning, landscaping and stormwater objectives for the site re-development. Figs. 3 and 6 show a typical section of this multi-purpose integration for the local streets.

Other purposes included:

- to function as a drainage conveyance system for offsite and site stormwater;
- to function as stormwater retarding system with the aim to reduce stormwater contribution to the downstream stormwater conveyance system during large storm events;

Figure 6: Photo of a local street.

- to function as a water quality control system that will detain and filter pollutants from the stormwater. Controlling and trapping stormwater pollutants at their source has their advantages of reduced hydraulic loading, attenuate flows, reduce pollutant loads to downstream treatment facilities (i.e. drainage pipe, wetlands) and, in may cases lower capital costs;
- the bio-retention/infiltration system and wetlands with immediate/adjacent planting, will function as, habitat creation, aesthetics and create an environmental friendly environment.

21.9.3. Bio-retention/Infiltration System Details

Figs. 7–9 show the bio-retention/infiltration system details.

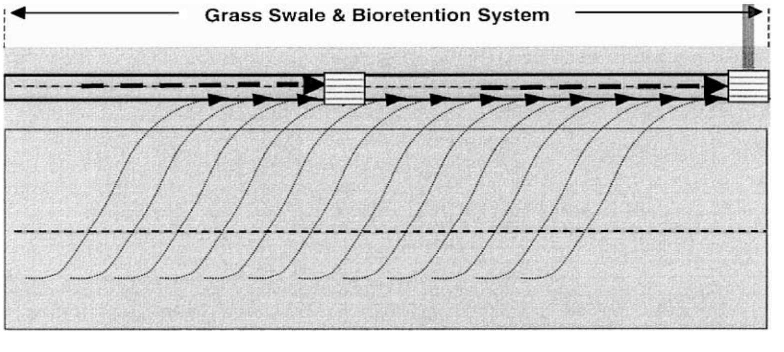


Figure 7: Plan view.

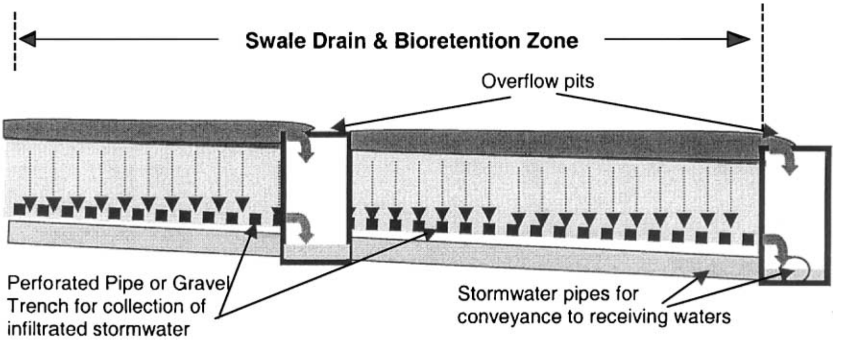


Figure 8: Section view.

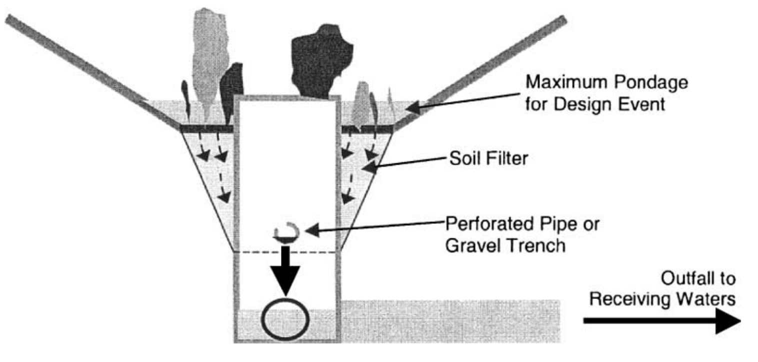


Figure 9: Cross-sectional view.

21.9.4. Bio-retention/Infiltration Variations

Figs. 10–12 show other variations of the bio-retention/infiltration system used by DPWS.

21.10. Management

To maximise pollutant capture/treatment, wetland ecosystems will need regular maintenance. The essential operation and management elements required for wetland ecosystems include: *a description of the management strategy* (detailing its objectives, functions and the relationship between the physical structures and

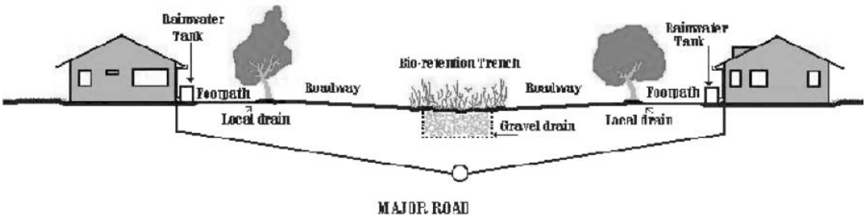


Figure 10: Major Road.

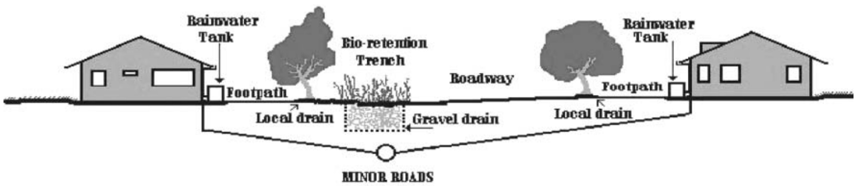


Figure 11: Minor Road.

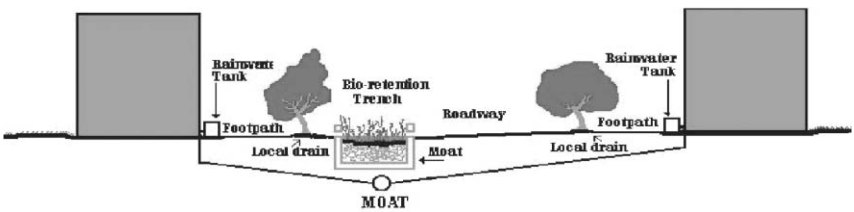


Figure 12: Moat.

the biological components); *a list of tasks or management activities*; *a management calendar* (to ensure that programmed maintenance activities are carried out); *monitoring activities* (with inspection checklists to ascertain that all components in a wetland cell are functioning properly); *safety measures* (to ensure that the wetlands are safe to visit and to work in); and *timelines* so that all the operation/works must be carried out on the specified time and within the period provided (Gan, 2001b; Gan and Beharrell, 2000).

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I wish to thank DPWS and our Clients for permission to use the information presented in this chapter.

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Chapter 22

The Application of Constructed Wetlands for Water Quality Improvement in the Deep Bay Catchment of Hong Kong

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Abstract. A mosaic of natural and man-made wetland habitats can be found in the northwestern New Territories of the Hong Kong SAR, supporting a high diversity of fauna and flora. The Inner Deep Bay mudflats is the core area of the Deep Bay wetland ecosystem, which provides feeding grounds for the tens of thousands of migratory waterbirds that annually stop-off in Hong Kong. Due to the rapid urbanization and population growth of the Deep Bay catchment, the water quality of Deep Bay is poor and consistently beyond the set objective. In order to protect the water environment and thus the wetland ecosystem of Deep Bay, and allow its assimilative capacity to be met in the long term, the pollution load entering the Bay needs to be significantly reduced. To achieve this, the Shenzhen SEZ and Hong Kong SAR Governments have formulated a joint implementation program to tackle Deep Bay's water quality problems. Stringent requirements for developments within the Deep Bay catchment have also been established. Developments in areas soon, or unlikely ever, to be connected to the public sewer have to provide effective onsite sewage treatment facilities and further measures to offset the residual pollution load from the facilities during the interim, or the life of the development. This is to achieve the requirement of no net increase in pollution loading into Deep Bay. The idea of using constructed wetlands to polish effluent from treatment plants or polluted streams in order to balance the net increase of pollution load is commonly adopted in planning proposals. So far a number of such proposals have been endorsed and some developments in the Deep Bay area are described. The effectiveness of constructed wetlands for water quality improvement in Hong Kong still needs to be demonstrated.

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22.1. Introduction

Constructed wetlands are man-made complexes consisting of different types of substrate, emergent and/or submerged vegetation, wildlife and water that simulate natural wetlands to treat wastewater. The processes of biological degradation, filtration, sedimentation and absorption result in significant reduction of suspended solids and organic pollutants. Such constructed wetlands are in widespread use in the USA and are becoming common in Europe, Australia and parts of Asia and Africa (Bastian & Hammer, 1993; Cooper & Green, 1995; Urbanc-Bercic & Bulc, 1995; Cooper et al., 1996). Interest has also grown in making use of wetland processes to treat urban and agricultural stormwater runoff (Raisin & Mitchell, 1995; Cooper et al., 1996). However, there is limited data and experience in Hong Kong on using constructed wetland for water improvement, apart from a mini setup at the Kadoorie Farm and Botanic Garden to demonstrate the function of constructed wetlands on treating wastewater from a livestock source. In the recent years, the approach of using constructed wetland has been widely proposed by development projects within the Deep Bay catchment in response to the stringent pollution control requirement imposed in Deep Bay for protecting the ecological integrity of that site (Town Planning Board, 1999). This chapter reviews the status and strategy of a few ongoing projects on using constructed wetland to achieve the planning requirements.

22.2. Deep Bay Catchment

Deep Bay is a large shallow semi-enclosed bay located in the northwestern New Territories of Hong Kong. It is fed by water from the Pearl River Estuary and several rivers around the bay (Fig. 1). The total surface area of the Bay is approximately 112 km, with a length of about 15 km and an average depth of 3 m. The total catchment area of the Bay covers about 535 km^2, of which 51% lies on the Shenzhen side and 49% lies in the New Territories of Hong Kong (Environmental Protection Department, 1998). The Shenzhen River flows from the northeast to southwest into Deep Bay and forms the boundary between Hong Kong and Shenzhen.

The Deep Bay catchment comprises many high-valued ecological sites, especially around Inner Deep Bay. On the Hong Kong side of this site, different natural and man-made wetlands provide a wide range of habitats to support a high diversity of biota. Habitats including the mudflats, mangroves, traditional shrimp ponds (*gei wai*), inundated marshlands and extensive man-made fishponds in these areas are recognized to have high ecological values, and form a complex wetland

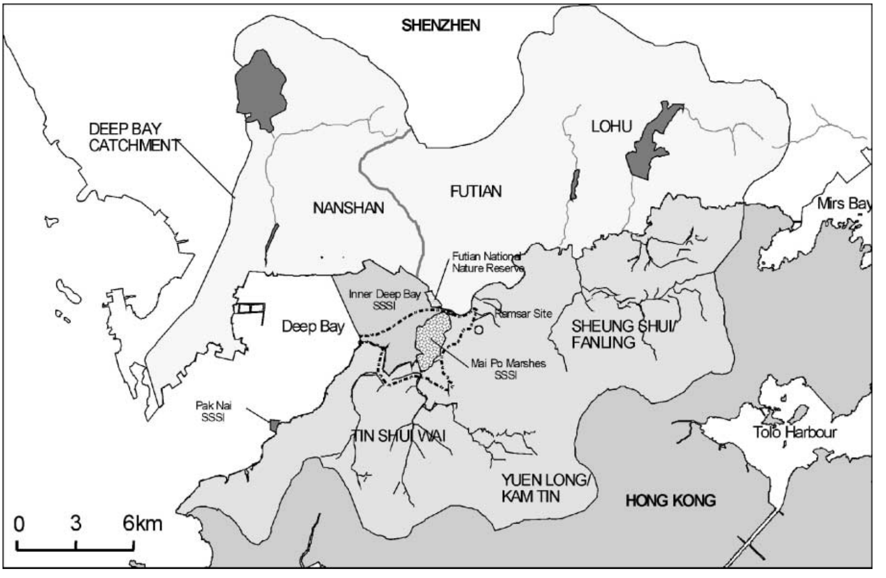

Figure 1: Delineation of the Deep Bay catchment and the distribution of the major sensitive sites around Deep Bay.

ecosystem in Inner Deep Bay (Planning Department, 1997). The wetlands within and surrounding the Mai Po Nature Reserve are an especially important roosting and feeding ground for ten of thousands of migratory birds and variety of local species. A 1,500 ha area of the Mai Po Marshes, Inner Deep Bay and surrounding fishponds were listed as a "Wetland of International Importance" (Ramsar Site) in 1995. According to the SSSI registration and the latest review (Planning Department, 1999), there are a number of important ecological sensitive receivers around the coast of and at Deep Bay. These include the Inner Deep Bay Site of Special Scientific Interest (SSSI) (1986), Futian National Nature Reserve in Shenzhen, Mai Po Inner Deep Bay Ramsar Site (1995), Mai Po Marshes SSSI (1976), Mai Po village SSSI (1979), Tsim Bei Tsui SSSI (1985), Tsim Bei Tsui egretry SSSI (1989), Pak Nai SSSI (1980), seagrass and horseshoe crab habitats along the coast of Pak Nai, and mariculture subzone along the coast of Lau Fau Shan and Pak Nai (Fig. 1).

22.3. Strategies on Water Pollution Control in Deep Bay

Due to rapid urbanization and development around Deep Bay, substantial amounts of pollutants are carried from the urban, industrial and rural areas of Hong Kong and Shenzhen into Deep Bay. The water quality of the Bay is consistently poor, especially in Inner Deep Bay, although substantial efforts have been spent since the 1980s to reduce pollution (Environmental Protection Department, 2000). A comprehensive study using computer modeling, was conducted between 1995 and 1998 to assess the assimilative capacity of the Bay (Environmental Protection Department, 1998; Lee, 2000). The results of the study showed that the assimilative capacity of the Bay has been far exceeded, especially for a few important water quality parameters, e.g. oxygen demand, nitrogen, phosphorus and bacteria. Apart from the existing and planned effluent treatment and export schemes in both the Hong Kong and Shenzhen sides of the catchment, further reduction in the loads discharged from the substantial population and associated commercial, industrial and livestock rearing activities are required. Accordingly, further reviews were undertaken in 1999 by the governments of Hong Kong and Shenzhen to identify areas for further reduction. A joint implementation program was devised outlining steps to achieve the target of pollution reduction within the next 15 years (HKGEPLG, 1999).

In the meantime, the Hong Kong SAR Government's Planning Department has established a new strategy to tackle the increasing development pressure in the Deep Bay area, such as the low-density residential developments (Fig. 2). To avoid

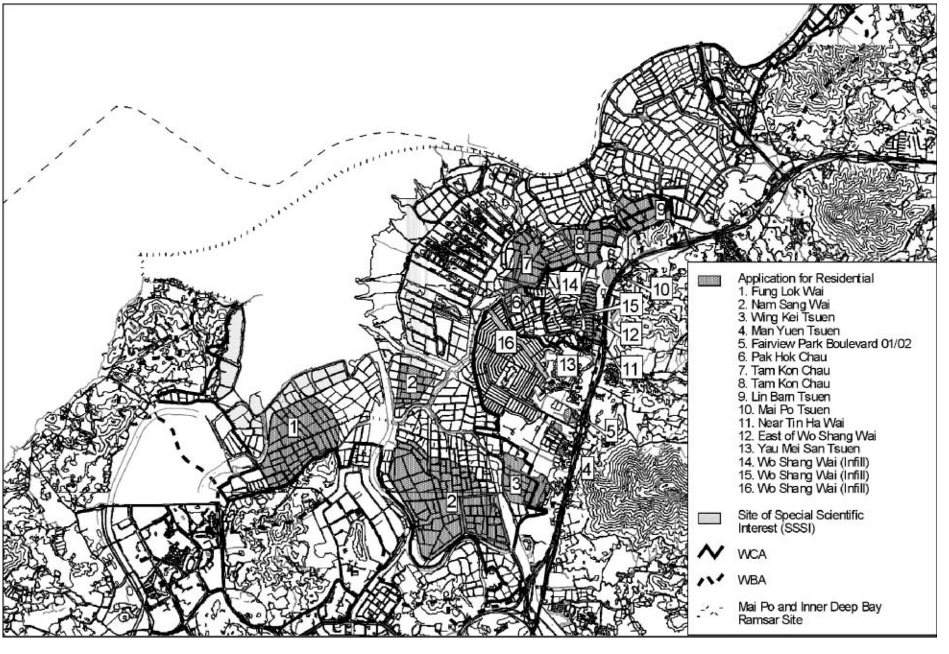

Figure 2: Delineation of the Ramsar site. Wetland conservation area (WCA) and wetland buffer area (WBA) and distribution of the residential proposals in the sensitive zone.

any adverse impact on the Deep Bay Wetland ecosystem, the Town Planning Board has designated the landward part of the Ramsar site and its surrounding continuous and adjoining fishponds, as a wetland conservation area (WCA). A strip of land about 500 m wide along the landward side of the WCA was further designated as a wetland buffer area (WBA) to protect the ecological integrity of the WCA (Town Planning Board, 1999) (Fig. 2). According to the planning guidelines, any developments within the WCA and WBA should fulfill the requirement of no net increase of pollution load to Deep Bay. To achieve this, mitigation measures have to be provided to offset any residual pollution loads from the development.

22.4. Application of Constructed Wetlands in the Deep Bay Catchment

22.4.1. Constructed Wetlands for Stormwater Runoff

Wetlands are transitional ecosystems that exist at the interface between aquatic and terrestrial systems. Because of their position in the landscape, they are frequently the default recipients of stormwater runoff. Increasing urbanization has led to large increase in the pollutant loads delivered to natural receiving waters. The use of constructed wetland for cleaning up stormwater runoff before discharging is widely adopted in order to protect the water environment (Raisin & Mitchell, 1995; Cooper et al., 1996; Godrej et al., 1999; IWA, 2000), but this arrangement is not common in Hong Kong. This is likely due to the limited resources for establishing constructed wetland and handling the issues of stormwater runoff in Hong Kong. Normally, standard practices are adopted to reduce the impact from stormwater runoff in urban or less sensitive areas, e.g. silt trap or grease trap. However, more stringent measures are necessary for developments within the Deep Bay catchment. This step is considered necessary as recent research has demonstrated that the heavy metal levels recorded in the feathers of the egrets breeding in colonies around Deep Bay were particularly high, especially for lead. Further additional input may cause biological impacts to the biota involved (Connell et al., 2000). To address this concern, one recent highway and bridge project has proposed the "control at sources strategy" to tackle the highway runoff problem (Highway Department, 2002). The proposed high frequency of road cleaning by vacuum sweeper is targeted to reduce pollutants accumulating on the bridge and risk of runoff into Deep Bay during storm events. The effectiveness of this measure will be monitored during the initial operation stage from around 2005.

In the 1990s, the large-scale Tin Shui Wai residential development in the northwest New Territories drew much attention from the public and green groups. The development site covered around 220 ha and would have a final total population of around 340,000 (Territory Development Department, 1997). The northeastern reach of the site was adjacent to the ecologically sensitive Ramsar Site (Fig. 3). Although the development would be connected to the public sewer, the potential impact of stormwater runoff was still a substantial environmental concern. To reduce the impact of stormwater runoff and avoid the interfacing problems between areas for people and conservation, the Tin Shui Wai development EIA recommended the creation of a constructed wetland to serve as a buffer to separate the Tin Shui Wai residential zone and the Ramsar Site. The proposed constructed wetlands, which covers 56 ha was aimed to provide opportunities for mitigation of wetland habitat loss due to the Tin Shui Wai development and for cleaning up part of the stormwater before it is discharged into Deep Bay. This constructed wetland then became the major part of the current Wetland Park project. Apart from having a function to improve water quality, the Wetland Park can also promote recreation, education and tourism (Agriculture, Fisheries and Conservation Department, 1999).

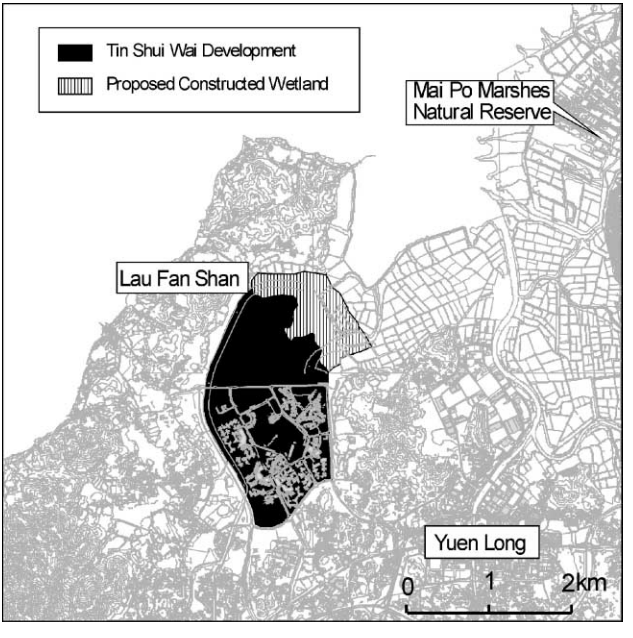

Figure 3: Location of the Tin Shui Wai development and the proposed constructed wetland.

To achieve the water improvement function, the Wetland Park incorporates a numbers of long channels with varying depths between 0.5 and 2.0 m and a series of shallow and deep water pools separated by weirs for flow control. A filter system composed of different type of wetland plants, e.g. reed bed and sedge, are integrated into the overall wetland design to provide a cleanup function for a portion of the stormwater flowing from the development area. Water quality monitoring at various locations within the constructed wetland is specified to monitor the treatment efficiency of the water circulation system. Apart from the general in situ parameters, ammonium-N, total phosphorous, biological oxygen demand, Chlorophyll-a and suspended solids will be monitored. After the first year of monitoring, the program will be reviewed to assess the effectiveness of the constructed wetland system for water improvement and the adequacy of the monitoring program. Some initial data have already collected and the system will soon be in operation.

22.4.2. Construction Wetlands for Polishing Treatment Plant Effluents and Stream Water

Development pressure in the northern New Territories is very high, especially from low-density residential developments which occur around the Deep Bay wetland area (Fig. 2). This area is largely rural and unsewered, although a trunk sewer is planned to be in operation around 2007. According to the Deep Bay planning guidelines, any developments that take place within this area before the trunk sewer is in place, will have to provide effective sewage treatment facilities and offsetting measures to achieve the requirement of no net increase of pollution load into Deep Bay. Generally, secondary treatment plus disinfection is the proposed treatment level, whilst the offsetting measures mainly include the proposal for reusing treated effluent for toilet flushing and gardening, pumping the equivalent amount of polluted stream water into the plant for treatment, and using constructed wetlands onsite or offsite to treat the effluents and/or water from the polluted streams. At the moment, many such residential proposals are either at the stage of planning, application process or pending approval, e.g. the developments at Nam Sang Wai and Fung Lok Wai. Due to the sluggish local investment market and economic downturn in recent years, many residential development projects are either on-hold or have been abandoned.

Apart from residential projects, one re-development project and one infrastructure project, respectively, are, however, approaching the stage of operation and detail design. These are the *Expansion of Kiosks at Lok Ma Chau Boundary Crossing (Boundary Crossing Project)* and *Sheung Shui to Lok Ma Chau Spurline (Spur Line Project)* (Fig. 4). Both of these projects lie within

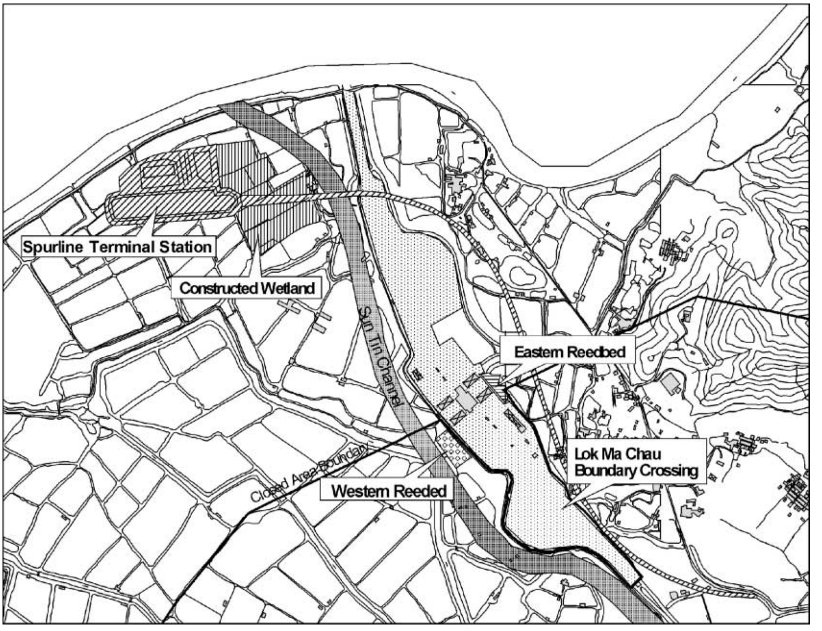

Figure 4: Location of the constructed wetlands proposed by the Lok Ma Chau Spur Line and boundary crossing projects.

the WCA and unsewered zone. Accordingly, they have to provide onsite sewage treatment facilities and fulfill the offsetting requirements. The offsetting plans of these two projects are briefly reported below:

Boundary Crossing Project. The Lok Ma Chau boundary crossing is one of the three crossings between Hong Kong and Mainland China. It was opened in 1989 with a total of 14 pairs of customs kiosks installed. The design capacity of the crossing is 1,000 vehicles per hour while the present peak traffic flow has risen to around 1,400–1,500 per hour, greatly exceeding the design capacity. With the increase in the freight and passenger traffic set to continue, it was proposed that 10 additional pairs of kiosks and other facilities be added to the crossing point (Architectural Services Department, 1999). This arrangement will directly affect the number of people crossing the border, and thus the amount of sewage production at the crossing. The original septic tank and soakaway system is insufficient to cater for the average of 25,000 passengers per day, with the sewage flow around 80 m^3/day. Accordingly, it was proposed that an onsite rotating biological contactor (RBC) sewage treatment plant be constructed to achieve the discharge standards (Table 1) (Environmental Protection Department, 1991), with

Table 1: Effluent standards of the related parameters for discharging into the "group B inland waters", which represent the beneficial use for irrigation.

Parameters	**Discharge standard based on effluent flow rate (m^3/day)**	
	$\leq$ 200	**>1000 and $\leq$ 1500**
pH	6.5–8.5	6.5–8.5
BOD_5 (mg/l)	20	20
COD (mg/l)	80	80
Suspended solids (mg/l)	30	30
Ammonia nitrogen (mg/l)	5	5
Nitrate + nitrite (mg/l)	30	20
Total phosphorous (mg/l)	10	8
E. coli (count/100 ml)	100	100

the mitigation measures to offset the residual pollutant load that would be discharged into Deep Bay from the RBC treatment plant.

Two patches of constructed reedbeds are proposed for the offsetting plan (Kowloon-Canton Railway Corporation, 2000) (Fig. 4). The eastern reedbed will receive a constant flow of effluent from the RBC, whereas the western reedbed will treat water extracted from the nearby San Tin Channel as the external pollutant source for compensation of the load discharged from the RBC. As the residual load from the RBC can be calculated and the purification function of the reedbed can be estimated through regular monitoring at the input and output points, the effectiveness of the offsetting measures can thus be assessed. For example, according to the RBC design, the BOD_5 concentration in its effluent can achieve the level around 15 mg/l. According to the estimated daily sewage flow, the annual load from the RBC is thus around 1,116 kg, which needs to be offset by both the eastern and western reedbeds. Assuming the eastern reed bed achieves 50% removal of BOD_5 from the constant effluent from RBC, the western reed bed has to treat the remaining 50%, i.e. 583 kg/l BOD_5 via extracting enough water from the San Tin Channel to treat that equivalent amount of BOD_5.

Due to the lack of information on the general performance of constructed wetlands in Hong Kong, it is considered inappropriate to set the daily compliance of offsetting at this stage. Accordingly, project proponents are requested to achieve only the annual balance between the residual load from the RBC and removal by the reedbed system. BOD_5 is the parameter used to assess the effectiveness of the reedbed system for the offsetting, although other parameters will also be monitored on a regular basis (Table 2). The monitoring results and case review

Table 2: Parameters to be measured in water, sediment and plant samples collected from the constructed wetlands.

Parameters in water samples	
In situ	Flow rate, temperature, salinity, pH, turbidity, dissolved oxygen
Nutrient	BOD_5, suspended solids, total phosphorus, ammoniacal nitrogen, nitrite, nitrate, total Kjeldahl nitrogen, TOC, COD, *E. coli*
Metal	Cadmium, lead, copper, zinc, iron, arsenic
Parameters in sediment and plant samples	
Nutrient	Organic nitrogen, total Kjeldahl nitrogen, total nitrogen, organic phosphate, total phosphate
Metal	Cadmium, lead, copper, zinc, iron, arsenic

will provide a basis for formulating the offsetting compliance requirement in future. At present, the eastern reedbed is under construction.

Spur Line Project. This Spur Line Project consists of the construction and operation of a railway station at Lok Ma Chau, and railway alignment between Sheung Shui and Lok Ma Chau (Kowloon-Canton Railway Corporation, 2000) (Fig. 4). The project aims to relieve the congestion at Lo Wu Boundary Crossing. It was estimated that the peak flow of passengers to cross this new boundary crossing might reach around 20,000 passengers per hour. Including the approximately 1,100 staff at the Lok Ma Chau station, the estimated maximum daily sewage flow is around 1,228 m^3/day. As the Lok Ma Chau station lies within the WCA and unsewered area, the construction of a RBC treatment plant with a disinfection setup was proposed to achieve the discharged standards (Table 1). The effluents will be further polished by a 2 ha constructed reedbed. In the meantime, water from the San Tin Channel will be pumped to the reedbed for treatment in order to offset the residual pollution load from the treatment plant. This strategy is similar to that adopted at the nearby Boundary Crossing Project. Again, similar monitoring will be carried out during the operation phase to assess the effectiveness of the reedbed system. The Spur Line Project is now at the stage of detail design and construction. Its operation phase will begin around 2006.

22.5. Conclusion

The Deep Bay catchment has a high potential to promote the use of constructed wetlands for water improvement. With the benefit of less limitation on land resources, the constructed wetland can fit well into the original landscape that supports a wide range of wetlands. The current ongoing large-scale projects described above will help to collect useful information to verify the effectiveness of constructed wetland for water improvement in Hong Kong. If the results demonstrate that the approach of constructed wetland is viable and satisfactory, it will further help to improve the polluted water environment of Deep Bay.

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Chapter 23

Use of a Wetland System for Treating Pb/Zn Mine Effluent: A Case Study in Southern China from 1984 to 2002

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Abstract. A constructed wetland system in Guangdong Province, South of China has been used for treating Pb/Zn mine discharge since 1984. In this chapter, the performance of this system in the purification of mine discharge, metal accumulation in different ecological compartments and ecological succession within the system during the period of 1984–2002 has been reviewed. The data show that the wetland system not only effectively remove metals (mainly Pb, Zn, Cd and Cu) and total suspended solids from the mine discharge over a long period leading to significant improvement in water quality, but also gradually increase diversity and abundance of living organisms.

23.1. Introduction

Water quality problems can be caused by water storage and by pollution due to effluent discharge to the drainage system (Finlayson & Mitchell, 1982). The deliberate use of wetlands (both natural and constructed) as biological treatment systems for effluent purification has developed rapidly over the last 30 years with the increasing scientific documentation of the role of plants in wastewater purification (Wolverton, 1987; Dunbabin & Bowmer, 1992; Sundaravadivel & Vigneswaran, 2001). The growing interest in wetland systems is in part due to the recognition that natural treatment systems offer advantages over conventional concrete-and-steel, equipment-intensive, mechanical treatment plants. When the same biochemical and physical processes occur in a more natural environment instead of reactor tanks and basins, the wetland system often consumes less energy, is more reliable, requires less operation and maintenance and, as a result, costs less (Smith, 1989). Most research on the use of wetlands for wastewater

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treatment has been directed towards using municipal wastewaters to reduce the concentrations of nitrogen and phosphorus and to lower the biological oxygen demand. The use of wetlands to treat metal-contaminated mine drainage water by removing metals has been largely ignored until the last decade (Dunbabin & Bowmer, 1992).

In comparison with organic effluents, drainage waters from mining activities and abandoned mines are frequently higher in metal loads, more poorly buffered, of more extreme pH, lower in organic compounds and fluctuate in volume. Metals present in mine discharges, such as lead, zinc, cadmium and copper can pose serious environmental threats, as they are potentially toxic to all living organisms. Metal-contaminated mine drainage waters cause widespread and serious water pollution problems as they can ruin fisheries and recreational lakes, damage structures, increase the cost of municipal water treatment, degrade the value of land and lower the potential of an area for tourism (National Rivers Authority, 1992; Sundaravadivel & Vigneswaran, 2001).

It is both a difficult and expensive task to decontaminate large volumes of wastewater polluted with heavy metals by conventional physico-chemical techniques. As Eger et al. (1994) have indicated, although this type of drainage can be chemically treated in an active treatment plant, this is an expensive and long-term commitment, particularly since drainage problems can persist for over a 100 years. In western countries some successful case studies have indicated that wetlands can effectively purify metal-contaminated mine drainage water, and can offer an economical, self-maintaining, and therefore preferred alternative to conventional treatment of different types of contaminated water (Gersberg et al., 1984; Erten et al., 1988; Hammar & Bastian, 1989; Wildeman & Laudon, 1989; National River Authority, 1992; Ye et al., 2001a,b).

In P.R. China, the drainage water from mining activities has caused some serious environmental problems, but little information is available concerning the use of wetlands to treat metal-contaminated mine drainage water. In this chapter, some experimental results are reviewed for a wetland system dominated by *Typha latifolia* (cattail) to treat the wastewater generated from the lead/zinc (Pb/Zn) mine at Shaoguan, Guangdong Province, China, during the period of 1984–2002. Full experimental details can be found in the various papers referred to.

23.2. Description of the Study Site

The Pb/Zn mine is situated at Shaoguan, Guangdong, in the subtropical region of China. The mean annual temperature is about 20°C and the extreme values are −5°C in January and 40°C in July. Average annual rainfall is 1,457 mm. The total land area of the mine is about 4 km^2, and it is situated at 100–150 m above sea

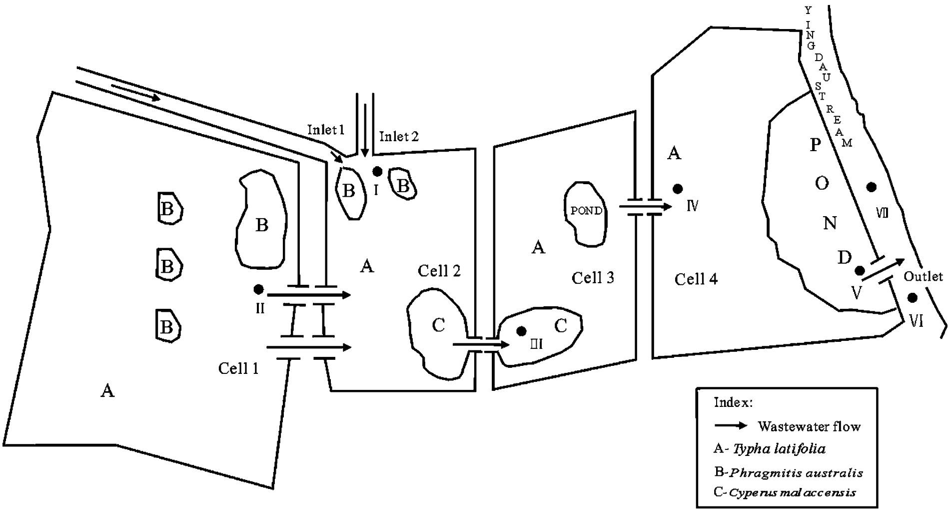

Figure 1: Sketch map of Fankou wetland system.

level. The original area of the wetland system (Cell 1) (Fig. 1) was about 87,500 m^2; it was 350 m in length and 250 m in width with an average depth of 2.5 m. The system was enclosed by dam walls constructed of rocks and mine tailings. Its capacity was about 150,000 m^3, and the treatment capacity was about 29,800 t day^{-1}, with a retention time in the range of five to seven days. This system could be divided into two parts: an upper part that was a wetland dominated by cattail, and a lower, a stabilization pond without physical separation. The mean depths of the wastewater in the wetland area and the stabilization pond were about 10 cm and 2.5 m, respectively. There were two entrances to the combined treatment system. Cattail was planted in the wetland area in 1983, and after two years it had expanded to cover about 55% of the area; after six years (1989), it had colonized about 73.6% of the area. No hydrophytes had been found on the stabilization pond. The wastewater draining from the entrances flowed into the aquatic treatment system and then into the stabilization pond, finally leaving through two exits. About 26,000 tonnes drainage per day were discharged into the system during the period of 1984–1989. Because of the high content of total suspended solids (TSS) in the drainage; an average of about 52,900 t per year of sediment were deposited in the system. This sediment gradually raised the bottom of the system thus decreasing both volume and treatment capacities, therefore, Cells 2–4 were constructed in 1988, 1995 and 1998, respectively (Fig. 1, Table 1). Cattail is the major dominant plant species in all four cells, other dominant species included *Phragmites australis* and *Cyperus malaccensis*. Cell 1 and Cell 2 ceased to be used in treating mine drainage in 1995 and October 2001, respectively.

Table 1: The time schedule of Fankou wetland cells used for treating Pb/Zn mine drainage in Shaoguan, Guangdong Province, Southern China.

	Cell 1	Cell 2	Cell 3	Cell 4
Year wastewater treatment started	1984	1988	1995	1998
Year wastewater treatment finished	1995	2001	–	–
Area (m^2) of cell	87,500	23,510	14,950	33,220

Although mine drainage had not been discharged into Cell 1 since 1995, metal polluted-runoff from the surrounding area still passed through Cell 1 into Cell 2. The Cell 2 served as a ditch after its use for drainage treatment was stopped in October 2001.

23.3. Efficiency of Metal Removal from the Effluent by the Wetland System

Table 2 illustrates the changes in water quality resulting from the system by comparing the properties of the influent and effluent waters. The results indicate that influent water had a rather high level of TSS (4,635.4 mg l^{-1}), high levels of Pb (1.61 mg l^{-1}) and Zn (1.96 mg l^{-1}) and a moderate level of Cd (0.022 mg l^{-1}). After treatment by the system, water quality was significantly improved: pH changed from 8.03 to 7.78, TSS in the influent water were reduced by 99% ($p < 0.001$), Pb by 90% ($p < 0.001$), Zn 84% ($p < 0.001$), Cd 86% ($p < 0.001$), Fe 97% ($p < 0.05$). Other metals: Cu, Ca, Mg, Al, Na, Co and K were reduced by between 50 and 91% of their original concentrations in the influent water. The concentrations of soluble Pb, Zn, Cd, as well as other metals and TSS in the treated effluent, were all within the upper limits set for industrial wastewater discharge in China (Ye et al., 1992a).

In 1984, the mean Pb concentration in effluent water was 0.60 ± 0.147 (mg l^{-1}), and Zn: 1.09 ± 0.222 (mg l^{-1}), and pH: 8.07 ± 0.05 ($n = 15$, mean $\pm$ se, January–July 1984). The efficiencies of Pb and Zn removal from January to July 1984 were 62.7 and 44.4% after the drainage passed the same area but only a few patches of cattail grew within in the this area. The efficiencies of metal and TSS removal were increased rapidly after 1985 and tended to improve in subsequent years with increasing plant area in the system (Ye et al., 1992a).

General properties of influent and effluent waters were monitored again in 1998 and in period of June 2001–May 2002. The data presented in Table 3 show that TSS in the influent water were reduced by 99.6%, Pb by 80.1%, Zn 97.2%,

Table 2: General properties of influent and effluent water collected at the entrance and exit of the purification system compared with the industrial discharge standard (mean values ± se) (January 1985–December 1989).

Property	Influent	*n*	Effluent	*n*	Industrial discharge standard
pH	8.03 ± 0.08	7	7.78 ± 0.03	174	
Total suspended solids ($mg\ l^{-1}$)[a]	4635.40 ± 384.49	5	28.11 ± 10.99	5	500
Pb ($mg\ l^{-1}$)[a]	1.61 ± 0.41	21	0.157 ± 0.019	177	1.00
Zn ($mg\ l^{-1}$)[a]	1.96 ± 0.41	23	0.309 ± 0.026	177	5.00
Cd ($mg\ l^{-1}$)[a]	0.022 ± 0.005	7	0.003 ± 0.0002	136	0.10
Cu ($mg\ l^{-1}$)	0.044 ± 0.012	7	0.017 ± 0.005	7	
Fe ($mg\ l^{-1}$)[a]	17.26 ± 6.26	7	0.49 ± 0.17	7	
Al ($mg\ l^{-1}$)	12.43	3	0.002	3	
Ca ($mg\ l^{-1}$)	195.71	3	41.25	3	
Co ($mg\ l^{-1}$)	0.013	3	0.001	3	
K ($mg\ l^{-1}$)	5.00	3	2.50	3	
Mg ($mg\ l^{-1}$)	112.86	3	33.75	3	
Na ($mg\ l^{-1}$)	8.86	3	3.88	3	

Ye et al. (1992a). *n* denotes the number of samples.
[a] *t*-test indicate a statistical difference of $p < 0.05$ between samples values of influent and effluent water.

Cd 96.5%, and Cu 96.8% after treatment by the system. The average concentration recorded in the influent water were: 99.3 $mg\ l^{-1}$ Pb, 61.4 $mg\ l^{-1}$ Zn (Inlet 1), 1.13 $mg\ l^{-1}$ Cd, and 1.15 $mg\ l^{-1}$ (Inlet 2); after treatment, however, only 0.128 $mg\ l^{-1}$ Pb, 0.34 $mg\ l^{-1}$ Zn, 0.003 $mg\ l^{-1}$ Cd and 0.008 $mg\ l^{-1}$ Cu were recorded in the effluent water (Table 3). The water quality of effluent was not only below the upper limits set for industrial wastewater discharge, but also met the Grade V quality standard for surface water: agricultural water for general landscapes (Table 4).

The data presented in Table 5 show the values for pH, TSS and metal concentrations in the surface water collected from site I (inlet of the system) to site V (outlet of the system), and from the upper and lower reaches of the receiving stream (sites VI and VII). The pH value was significantly increased from 6.48 in site I to 6.71 in site V, while concentrations of TSS, Pb, Zn, Cd and Cu in water were gradually reduced from the inlet to outlet of the system.

Table 3: General properties of influent and effluent water collected at the entrance and exit of the wetland system (mean values ± se, $n = 4$) in 1998, and June 2001–May 2002.

	Cell		pH	mS/cm	COD	TSS (mg l^{-1})	Pb (mg l^{-1})	Zn (mg l^{-1})	Cd (mg l^{-1})	Cu (mg l^{-1})
1998	2	Inlet	7.71	1.06	75.9	8802	2.41 ± 0.51	4.28 ± 1.20	0.80 ± 0.11	0.377 ± 0.044
	3	Outlet	7.34	0.49	7.69	31.5	0.48 ± 0.03	0.12 ± 0.08	0.028 ± 0.007	0.012 ± 0.004
		Reduction (%)			89.9	99.6	80.1	97.2	96.5	96.8
2001.6 ~ 2002.5	2	Inlet 1	7.34	3.10	–	1400	99.33 ± 13.7	61.4 ± 1.33	0.382 ± 0.226	0.756 ± 0.904
	2	Inlet 2	7.52	–	–	–	44.94 ± 5.78	27.5 ± 1.36	1.133 ± 1.530	1.15 ± 1.87
	4	Outlet	7.21	0.42	–	153	0.128 ± 0.03	0.34 ± 0.06	0.0026 ± 0.001	0.0076 ± 0.002

Yang et al. (2001) and Leung (2002).

Table 4: Chinese environmental quality standard for different grades of surface water.

	I	II	III	IV	V
pH			6–9		
Pb (mg l^{-1})	0.01	0.01	0.05	0.05	0.1
Zn (mg l^{-1})	0.05	1.0	1.0	2.0	2.0
Cd (mg l^{-1})	0.001	0.005	0.005	0.005	0.01
Cu (mg l^{-1})	0.01	1.0	1.0	1.0	1.0

Only pH and relative metal contents are listed in this table. China State Bureau of Environmental Protection Bureau (2002). According to the environmental functions and protective objectives of surface waters, all surface water in Mainland China are divided five grades: Grade I, source water or within national nature conservation zones; Grade II, surface water for drinking purpose (1st grade protection area), water for aquaculture (precious species); Grade III, surface water for drinking purpose (2nd grade protection area), water for aquaculture (common species); Grade IV, agriculture water for general landscapes.

The data in Table 5 also show that the outlet water from the system did not significantly increase metal concentrations in the receiving water body.

The above monitoring data show that the wetland system, whether consisting of one, two or three cells, was able to effectively remove metals and TSS from the mine drainage over a relatively long period. Reduction of TSS in the drainage before it was discharged into the system would elongate the life span of each cell.

23.4. Metal Accumulation in Different Ecological Compartments

23.4.1. Accumulation in Sediment

The data presented in Table 6 show that the sediment of the system (Cell 1) contained very high concentrations of Pb and Zn, and medium high concentration of As. Concentrations of these three metals in sediment at entrance site were about 66, 20 and 6 times higher than those in "clean" soil collected from control site (about 40 km west of Shaoguan Pb/Zn mine), respectively. Except for As, Cd, and Na, the other metal concentrations in the sediment were similar between the entrance site and the exit site of the system. Concentrations of As, Cd and

Table 5: Physical and chemical characteristics of surface water in January 2002 (mean ± se, $n = 3$).

Sites	**pH**	**TSS ($g\ l^{-1}$)**	**Pb ($mg\ l^{-1}$)**	**Zn ($mg\ l^{-1}$)**	**Cd ($mg\ l^{-1}$)**	**Cu ($mg\ l^{-1}$)**
I[a]	6.48 ± 0.01c[b]	0.22 ± 0.02a	44 ± 1.3a	17 ± 0.33a	0.02 ± 0.001a	0.03 ± 0.003a
II	6.62 ± 0.07bc	0.11 ± 0.03ab	0.02 ± 0.02e	0.27 ± 0.08c	0.006 ± 0.006b	0.04 ± 0.039a
III	6.54 ± 0.02bc	0.22 ± 0.01a	38 ± 1.8b	16 ± 0.58a	0.02 ± 0.001a	0.02 ± 0.001a
IV	6.84 ± 0.02a	0.18 ± 0.01ab	5.7 ± 0.67d	3.3 ± 1.3b	0.01 ± 0.001ab	0.01 ± 0.001a
V	6.71 ± 0.03ab	0.07 ± 0.001b	0.05 ± 0.03e	0.37 ± 0.21c	0.005 ± 0.001b	0.005 ± 0.002a
VI	6.62 ± 0.06bc	0.11 ± 0.04ab	0.07 ± 0.05e	0.06 ± 0.05c	0.0024 ± 0.0009b	0.019 ± 0.031a
VII	6.85 ± 0.04a	0.12 ± 0.03ab	0.05 ± 0.03e	0.11 ± 0.06c	0.0003 ± 0.0005b	0.018 ± 0.017a

Yu et al. (2004).

[a] I: Inlet of Cell 2, II: outlet of Cell 1, III: inlet of Cell 3, IV: inlet of Cell 4; V: outlet of Cell 4, VI: upper reach of the stream, VII: lower reach of the stream.

[b] Different letters in the same column indicate a significant difference at $p < 0.05$ according to Tukey-HSD test.

Table 6: Total concentrations of metals and N, P in the soil/sediment collected at the entrances and exits of wetland system and control site (samples collected from Cell 1 in period of December 1988–March 1990) (mg kg^{-1}, mean ± sd).

Order	**Sites**	**Total N (%) (n = 3)**	**Total P (%) (n = 3)**	**Pb (n = 7)**	**Zn (n = 7)**	**Cd (n = 7)**	**Cu (n = 7)**	**Fe (n = 5)**	**K (n = 5)**	**Na (n = 5)**	**Mn (n = 5)**	**As (n = 5)**	***t*-test of differences between sites**
1	Entrance	0.084	0.109	5,977 ± 2,191	3,057 ± 162	24 ± 7.0**	87 ± 16	35,974 ± 10,544	10,463 ± 4,023	1,257 ± 42**	1,549 ± 602	529 ± 162*	1–2
2	Exit	0.074	0.120	5,395 ± 2,457**	2,960 ± 420**	17 ± 9.1*	112 ± 32**	40,925 ± 6,723**	9,006 ± 3,413	943 ± 84**	1,454 ± 372	201 ± 177*	2–3
3	Control	0.174	0.047	91 ± 49**	153 ± 83**	1.52 ± 0.40**	22 ± 11**	18,008 ± 3,827*	7,419 ± 1,876	1,392 ± 77**	884 ± 582	86 ± 81**	1–3

Ye et al. (1992b). Probability values for t: *$p < 0.05$, **$p < 0.01$.

Table 7: Total concentrations of Pb, Zn, Cd and Cu in sediments of Fankou wetland system (samples collected in period of June 2001–January 2002, mg kg^{-1}, mean ± sd, $n = 9$).

	Pb	Zn	Cd	Cu
Cell 1	6,747 ± 3,228a[a]	5,697 ± 2,074a	15 ± 4.5a	145 ± 115a
Cell 2	5,124 ± 2,059a	4,117 ± 2,678a	12 ± 5.0a	70 ± 17a
Cell 3	4,538 ± 1,692a	4,960 ± 2,307a	12 ± 7.0a	84 ± 28a
Cell 4	3,992 ± 1,069a	4,611 ± 1,527a	14 ± 3.2a	69 ± 21a

Leung (2002).
[a] Different letters in a same column indicate a significant difference at $p < 0.05$ according to Tukey-HSD test.

Na in sediment were significantly higher in the entrance site than in the exit site (Ye et al., 1992b).

The data in Table 7 show the concentrations of Pb, Zn, Cd and Cu in the sediments of four wetland cells collected from June 2001 to January 2002. Although concentrations of Pb, Zn and Cu in sediment tended to decrease from Cell 1 to Cell 4, there were no significant differences between any of the cells. Compared to the data presented in Tables 6 and 7, the concentrations of Pb and Cu were similar, but Zn concentrations in the sediments were obviously higher in 2001–2002 than in 1988–1990. Concentration of total N in sediment of the wetland in Cell 1 was half that of the control soil, but the reverse was true for total P (Table 6).

23.4.2. Accumulation in Plants

The average concentrations of Pb, Zn and Cu in belowground tissues (roots, rhizome) of cattail were obviously higher than in aboveground tissues (Tables 8 and 9). Cattail grown in the system accumulated much higher concentrations of Pb and Zn than when grown in the control site (Table 8). Concentrations of Pb, Zn, Cd and Cu in both aboveground and belowground tissues of cattail grown in Cell 1 were similar to those of cattail grown in the other three cells (Table 9). The concentrations of metals in the aboveground tissues of cattail were similar in the two surveys carried out at different times. Both data suggest that cattail mainly excludes metals from its aboveground tissues and maintains low metal concentrations in its aboveground tissues, despite high metal concentrations in both sediment and its belowground tissues.

Table 8: Metal concentrations in root, rhizome and leaf of *Typha latifolia* grown in the Fankou wetland system and a control site (mg kg^{-1}, mean ± sd, $n = 7$).

Site	Organ	Pb	Zn	Cd	Cu	Mn
Fankou	Root	1,108 ± 693a[a]	946 ± 362a	1.5 ± 0.25a	29 ± 3.7a	178 ± 50b
Control	Root	90 ± 5.3b	139 ± 45b	1.3 ± 0.66a	10 ± 1.4b	531 ± 61a
Fankou	Rhizome	354 ± 182a	456 ± 175a	1.6 ± 1.6a	17 ± 6.5a	138 ± 19b
Control	Rhizome	39 ± 14b	78 ± 22b	0.81 ± 0.51a	4.8 ± 0.22b	335 ± 35a
Fankou	Leaf	99 ± 53a	155 ± 77a	0.62 ± 0.23a	9.1 ± 3.8a	586 ± 137a
Control	Leaf	15 ± 7.2b	43 ± 8.0b	0.55 ± 0.32a	3.2 ± 0.26b	664 ± 48a

Ye et al. (1992b).
[a] Different letters in a same organ and a same metal indicate a significant difference at $p < 0.05$ according to *t*-test.

23.4.3. Accumulation in Animals

Animal samples were collected in March and November 1990. Metal concentrations in muscle tissue varied greatly among the different species, for example, Pb contents ranged from 0.29 mg kg^{-1} in *Ophiocephalus maculates* to

Table 9: Concentrations of Pb, Zn, Cd and Cu in aboveground tissues and belowground tissues of *Typha latifolia* grown in the Fankou wetland system (samples were collected during the period June 2001–January 2002, mg kg^{-1}, mean ± sd, $n = 9$).

	Pb	Zn	Cd	Cu
Aboveground tissues				
Cell 1	118 ± 76a[a]	116 ± 46a	0.82 ± 0.46a	24 ± 17a
Cell 2	115 ± 96a	106 ± 41a	0.96 ± 0.84a	21 ± 10a
Cell 3	112 ± 76a	130 ± 60a	0.86 ± 0.60a	22 ± 15a
Cell 4	221 ± 124a	116 ± 70a	0.79 ± 0.91a	21 ± 21a
Belowground tissues				
Cell 1	4,108 ± 3,072a	2,641 ± 2,174a	3.4 ± 2.1a	32 ± 14a
Cell 2	1,315 ± 777a	749 ± 571a	2.8 ± 2.7a	25 ± 6.4a
Cell 3	1,456 ± 1,088a	1,125 ± 867a	6.7 ± 4.6a	23 ± 6.0a
Cell 4	1,497 ± 627a	1,271 ± 551a	8.4 ± 5.3a	21 ± 10a

[a] Different letters in a same column indicate a significant difference at $p < 0.05$ according to Tukey-HSD test.

Table 10: The concentrations of Pb, Zn, Cu and Cd in aquatic animal organism collected from the wetland system (mg kg^{-1}).

Species	**Sampling parts**	**Pb**	**Zn**	**Cu**	**Cd**
Cipangopaludina cathayensis	Muscle	68.5	169.2	15.3	1.48
Carassius auratus	Muscle	5.92	34.7	0.26	0.98
Ophiocephalus maculates	Muscle	0.29	8.87	1.84	0.08
	Liver	1.43	25.47	9.97	1.52
	Skeleton	27.1	107.5	4.05	0.65
	Egg	1.13	24.40	21.51	0.06
	Gill	31.4	63.1	4.14	0.26
	Scale	30.8	0.36	2.89	0.36
Maximum permitted concentration for consumption in China		1.0			0.5

Chen et al. (1990).

68.5 mg kg^{-1} in *Cipangopaludina cathayensis*, and Zn from 8.9 mg kg^{-1} in *O. maculates* to 169 mg kg^{-1} in *C. cathayensis* (Table 10). Among the three species tested, *C. cathayensis* accumulated the highest Pb, Zn, Cu and Cd in its muscle. Concentrations of Pb and Cd in muscles of *C. cathayensis* and *C. auratus* were higher than the maximum permitted concentration for consumption in China (PRC Agriculture Department, 2001), especially Pb in muscle of *C. cathayensis*, which was nearly 70 times higher than the maximum permitted concentration for this metal.

The concentrations of metals in different parts of the same animal body also varied greatly, for example, Pb ranged from 0.29 mg kg^{-1} in muscle to 31.4 mg kg^{-1} in gill tissue of *O. maculates*, and Zn from 0.36 mg kg^{-1} in scales to 107 mg kg^{-1} in skeleton of the same body. In *O. maculates*, muscle tissue accumulated the highest concentrations of Zn, scales the highest Pb (31.4 mg kg^{-1}), and egg the highest Cu (21.5 mg kg^{-1}) (Table 10).

23.5. Ecological Succession: Changes in Diversity and Abundance of Plants and Animals with Time and Space

23.5.1. Protozoa

Surveys of protozoa communities were conducted along the water flow from inlet to outlet of the wetland system in January 2002, using the polyurethane foam unit (PFU) method (China State Bureau of Technical Supervision and China EPA

Table 11: Protozoan communities in Fankou wetland system in 2002.

Sites	I	II	III	IV	V	VI	VII
Total number of protozoan species	0	28	0	1	28	65	62
Number species of Phytomastigophora	0	5	0	0	7	27	22
Percentage of Phytomastigophra	0	17.8	0	0	25.0	41.5	35.4

Yu et al. (2004).

1992). A total of 44 species was identified in water samples collected from the wetland system. No protozoa were observed in either sites I (inlet of the system) or III, and only one species was found in site IV. Twenty-eight species of protozoa, however, were found in site II (outlet of Cell 1) and site V (outlet of the system), respectively. There were 65 and 62 protozoan species recorded in the upper reach (site VI) and in the lower reach (site VII) of stream, respectively (Table 11).

The diversity of protozoa increased with the reduction of metals and TSS in the water (see Table 5). The higher diversity of protozoa in site II (28 species, outlet of Cell 1) may be due to the fact that Cell 1 had not been used in purifying mine drainage since 1995, so the water and the environment of this cell had improved gradually. Results of correlation analysis also indicate that both the species numbers and diversity index for protozoa were negatively correlated with concentrations of Pb, Zn, Cd and TSS in water (Table 12), which suggest that metals, especially Pb, Zn and Cd, and TSS in water play an important role in inhibiting the growth of protozoa, while the wetland system could effectively

Table 12: Relationships between biotic factors and abiotic factors in Fankou wetland system.

	Total species	Diversity index	Heterotrophic index
Pb	-0.70^{a}	-0.72^{a}	0.88^{a}
Zn	-0.74^{a}	-0.75^{a}	0.85^{a}
Cu	-0.07	-0.70	0.39
Cd	-0.70^{a}	-0.68^{a}	0.74^{a}
TSS	-0.71^{a}	-0.71^{a}	-0.73^{a}
pH	0.34	0.32	-0.77^{a}

Yu et al. (2004).
[a] Correlation coefficient, $p < 0.05$.

reduce the toxicity of the drainage and improve water quality, resulting in a higher microbial diversity.

23.5.2. Algae

Changes in Diversity with Time. The diversity of algae in the wetland system increased with time. In March 1986, only six genera belonging to three divisions were recorded in the system. Among these genera, three belonged to the Bacillariophyta, one to the Chlorophyta and two to Cyanophyta. *Nitzschia* was both highly abundant and also widely distributed in the system, including the heavy polluted sites (Table 13). Twenty-seven genera belonging to five divisions were recoded in March 1987. Among these 27 genera, 11 genera belonged to the Bacillariophyta, 8 to the Chlorophyta, 5 to the Cyanophyta, 2 to the Euglenophyta and 1 to the Cryptophyta (Table 14). The Bacillariophyta were widespread in the system with an abundance in genera, species and individuals, especially *Nitzcshia* (Table 15). A total of 40 algal species belonging to five divisions and 29 genera were observed in March 1989. Among these species, 17 belonged to 12 genera of the Bacillariophyta, 10 to 6 genera of the Chlorophyta, 9 to 9 genera of the Cynophyta, 3 to 1 genus of the Euglenophyta and 1 to 1 genus of Crytophyta. *Nitzchshia*, *Synedra* and *Oscillutouia* were dominant algae in the system, and these species were also distributed in heavily polluted areas within the system.

Changes in Diversity and Abundance with Space. The diversity and abundance of algae within the system increased from inlet to outlet (Tables 13 and 15). In March 1986, no *Nitzschia* were found in site I (inlet of the wetland), and only 48 (cells per liter water) in site II, however, numbers of this species increased rapidly along

Table 13: Algae (number of individual per liter) in wetland system (Cell 1) (samples collected in March 1986).

Site	*Nitzcshia*	*Fragilaria*	*Oscillutouia*	*Ocdogonium*
I[a]	0	0	0	0
II	48	0	0	0
III	2,676	191	96	95
IV	2,961	716	143	143
V	3,535	1,815	239	287

Chen et al. (1990).
[a] I: inlet of wetland 1, II and III: inside of Cell 1 along the water flow, IV: outlet of wetland, V: in the stream system nearby the wetland.

Table 14: Numbers of genus and individuals of algae in wetland system (samples collected in March and May 1987).

Division	**No. of genera**	**% of total genera**	**Individual (10^3 l^{-1})**	**% of total individuals**
Bacillariophyta	11	40.7	1,322.78	95.3
Chlorophyta	8	29.6	8.36	0.6
Cyanophyta	5	18.5	54.75	3.9
Cryptophyto	1	3.7	1.28	0.1
Euglenophyta	2	7.4	1.28	0.1
Total	27	100	1,388.45	100

Chen et al. (1990).

the flow, reaching 3535 in site V (outlet of the wetland). Similar trends were found in the other three alga species, *Fragilaria*, *Oscillutouia* and *Ocdogonium*. Similar results were reported in the survey conducted in 1990. The numbers of algal divisions and genera increased from 1 in site I to 5 and 15 in site IV, respectively. The number of diatoms was 50,000 (cells l^{-1}) in site I gradually increasing to 298,000 in site IV. No green algae and blue algae were found in site I, but 11,000 and 25,000 (cells l^{-1}) were recorded in site IV (Table 15). Only the diatom genus *Nitzchia* was found in site I. The above results show that the diversity and

Table 15: Algae in wetland system (Cell 1 and Cell 2) (samples collected in 1990).

Site	**Number of division**	**Number of genera**	**Number of individual per liter (cells l^{-1})**				**Number of *Nitzschia***	**Percentage of *Nitzchia* algae**
			Diatom	**Green algae**	**Blue algae**	**Total**		
I[a]	1	1	50,000	0	0	50,000	50,000	100
II	2	6	29,000	9,000	0	38,000	24,000	63
III	4	9	24,000	3,000	2,000	248,000	200,000	
IV	5	15	298,000	11,000	25,000	339,000	250,000	

Chen et al. (1990).

[a] I: inlet of Cell 1 (inlet of wetland), II: center of the Cell 1, III: outlet of Cell 1, and IV: outlet of Cell 2 (outlet of wetland).

abundance of algae gradually increased with water quality improvement or i.e. reduction of metal and TSS concentrations in water. Compared with the other algae, *Nitzcshia* showed higher tolerance to poor water quality and metal toxicity.

23.5.3. Higher Plants

During 1984 and 1989, the area of higher plants, mainly cattail, increased from about 100 m^2 to about 64,000 m^2 in Cell 1 (73.6% of Cell 1 in area). In addition to cattail, 11 other plant species were also found in 1989, including *Phragmites australis* and *Paspapum distichum*. Except above 12 plant species, four new species were found in the wetland in 1994. In 1998, 63 plant species belonged to 34 families and 59 genera were found within the wetland system (Table 16). The data from above three surveys show that plant diversity within the wetland system increased rapidly with time.

23.5.4. Benthic Invertebrates

Like the protozoa, no benthic invertebrates were found in site I (inlet of the system), diversity and abundance of these animals rapidly increased from site II to site IV, only 4 species were recorded in site II, but 8 and 9 species were found in site III and site IV, respectively (Table 17).

23.5.5. Vertebrates (Fishes, Terrestrial Animals and Birds)

Seven fish species were recorded in the stabilization pond in Cell 1 during 1986–1989. They were *Carassius auratus*, *Ctenopharyngodon idellus*, *Ophiocephalus maculates*, *Parasilurus asotus*, *Monopterus albus*, *Misgurnus anguillicaudaudatus* and *Oryzias latipes*.

Nine terrestrial animal species were found in the system in 1998 (Table 18). In the class Amphibia, *Bufo melanostiutus* and *Rana guentheri* were frequently found, while in the class Mammalia, *Rattus rattoides* was a dominant species.

Totally 26 bird species belonged to 7 orders and 13 families were observed in wetland with different degrees of abundance. *Ixobrychus cinnamomenus*, *Phylloscopus cantator ricketti*, *Orthotomus sutorius longicaudus*, *Prinia flaviventris delacouri*, and *Prinia subflava extensicauda* were dominant bird species in the system (Chang et al., 1999).

Table 16: The changes of plant species composition within the wetland system in period of 1989–1998.

Species composition	**1989**	**1994**	**1998**
Marchantiaceae			
Marchantia polymorpha L.			•
Leucobryaceae			
Leucobryum sp.			•
Equisetaceae			
Equisetum ramosissimum Desf.	•	•	•
Thelypteridaceae			
Cyclosorus acuminatus (Houtt.) Nakai			•
Ampelopteris prolifera (Petz.) Cop.			•
Marsileaceae			
Marsilea quadrifolia L.	•	•	•
Saururaceae			
Houttuynia cordata Thumb.			•
Papaveraceae			
Macleaya cordata (Willd.) R.Br			•
Fumariaceae			
Corydalis edulis Maxim.			•
Moraceae			
Ficus variolosa Lindl.			•
Urticaceae			
Boehmeria nivea (L.) Gaud.			•
Chenopodiaceae			
Spinacia oleracea L.			•
Chenopodium ambrosioides L.			•
Amaranthaceae			
Achyranthes aspera L.			•
Alternanthera philoxeroides (Mart.) Griseb.	•	•	•
Alternanthera sessilis (L.) Dc.			•
Caryophyllaceae			
Stellaria media (L.) Vill.			•
Polygonaceae			
Polygonum hydropiper L.	•	•	•
Polygonum chinense L.			•
Malvaceae			
Abelmoschus moschatus Medic.			•
Abutilon theophrasti Medic.			•
Malvastrum coromandelianum (L.) Garcke			•

(*continued*)

Table 16: Continued.

Species composition	1989	1994	1998
Cruciferae			
Capsella bursa-pastoris (L.) Medic.			•
Papillionaceae			
Pueraria phasedoides Benth.		•	•
Euphorbiaceae			
Bischofia polycarpa (Levl.) Airy	•	•	•
Euphorbia thymifolia L.			•
Sapium discolor (Champ.) Muell.-Arg.			•
Oxalidaceae			
Oxalis corymbosa DC.			•
Umbelliferae			
Centella asiatica (L.) Urban			•
Oenanthus benghalensis (Roxb.) Kurz.			•
Loganiaceae			
Buddleja asiatica Lou.		•	•
Solanaceae			
Solanum photeinocarpum Nak.et Odash.			•
Verbenaceae			
Verbena officinallis L.			•
Labiatae			
Prunella vulgaris L.			•
Perilla frutescens var.*acuta* (Thb.) Kudo			•
Callitrichaceae			
Callitriche stagnalis Scop.	•	•	•
Oleaceae			
Jasminum mesnyi Hance			•
Scrophulariaceae			
Mazus japonicus (Thb.) Kuntze			•
Compositae			
Ageratum conyzoides L.			•
Ambrosia artemisiifolia L.			•
Artemisia annua L.			•
Xanthium sibiricum Patr.			•
Youngia japonica (L.) Dc.			•
Hydrocharitaceae			
Hydrilla verticillata (L.f.) Royle			•
Potamogetonaceae			
Potamogeton crispus L.			•
Commeliaceae			

(*continued*)

Table 16: Continued.

Species composition	1989	1994	1998
Floscopa scandens Lour.			•
Commelina communis L.			•
Cyperaceae			
Fimbristylis sp.			•
Scirpus triqueter L.			•
Cyperus rotundus L.	•	•	•
Rhynchospora rubra (Lour.) Mak.			•
Gramineae			
Cynodon datylon L.	•	•	•
Paspalum distichum L.	•	•	•
Pennisetum purpureum Schum.			•
Imperata cylindrical var. *major* Habb. et Vs		•	•
Phragmites communis (L.) Trin.	•	•	•
Neyrandia reynaudiana (Kunth) Keng ex Hich.		•	•
Panicum repens L.			•
Digitaria chinensis (L.) Scop.			•
Mischanthus floridulus (Labill.) Warb.		•	•
Leersia hexandra Sw.	•	•	•
Typhacaeae			
Typha angustifolia L.			•
Typha latifolia L.	•	•	•

Yang et al. (2001).

23.6. Conclusions

1. The wetland system is able to effectively remove metals (mainly Pb, Zn, Cd and Cu) and TSS from the Pb/Zn mine drainage over a long-term period.
2. The wetland system not only improves the quality of the flow through drainage water, but also ameliorates the local environment by reducing tailings dust and erosion, improving the appearance of the landscape, and providing good habitats for different kinds of algae, plants and animals.
3. Diversity and abundance of living organisms in the wetland system were gradually increases with maturation of the system and improvement of water quality.
4. The system requires only low maintenance cost and is easy to manage, but needs larger areas of land than conventional methods. Reduction of TSS in the drainage before it is discharged into the wetland would elongate life span of each cell and reduce use of land.

Table 17: The main genera and species of benthic invertebrates and their distribution in the wetland system (samples were collected in March and November 1990).

Sites[a]	**I**	**II**	**III**	**IV**
Hydra sp.			++	
Nais variabilis		++		
Branchiodrilus hortensis		+		
Cipangopaludina cathayensis			++	++
Lymnaea stagnalis			++	+
Assiminea sp.			++	
Caridina nilotica gracilipes			+	
Caridina denticulate sinensis				+
Laccotrephes japonensis			+	
Naucoris exclamationis				+
Hyphydrus sp.				+
Chaoborus sp.				**+**
Tendipedidae		+	+	+
Tendipes plumosus		+		++
Clinatanypus sp.			+	
Tanytarsus sp.				+

" + " means the existence of the species; "++" means that the species appeared in a relatively higher numbers.
[a] Site I: inlet of wetland (Cell 1), II: inside of Cell 1, III: outlet of Cell 1, IV: outlet of wetland (Cell 2).

Table 18: Diversity of terrestrial animals within the wetland system.

Class	**Order**	**Family**	**Species**
Amphibia	Salientia	Bufonidae	*Bufo melanostiutus*
		Ranidae	*Rana adenopleura* Boulenger
			Rana guentheri Boulenger
			Rana limnocharis Boie
	Anura	Microhylidae	*Microhyla ornate* (Dumeril et Bibron)
			Microhyla pulchra (Hallowell)
Mammalia	Serpentiformes		*Natrix subminiata* helleri Schmidt
	Insectivora		*Suncus murinus* Linnaeus
	Rodentia	Muridae	*Mus musculus* Linnaeus
			Rattus rattoides Hodgson
			Rattus norvegicus Berkenhout

Hu (1998).

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Chapter 24

Wetland Creation in Hong Kong

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In nature, wetlands are formed only by natural forces over a relatively long period of time. Can humans mimic nature to recreate wetlands? Is wetland creation/restoration a way ahead for wetland conservation?

Abstract. Creation and restoration of wetlands have in recent years gained momentum over many parts of the globe. The trend has come to Hong Kong. Wetlands in Hong Kong are mostly created or restored simply as a result of development activity (e.g. residential developments, road and railway construction). In essence, a wetland is gained for each wetland lost. Many advocate that wetland mitigation is a promising approach in resolving the long-standing conflict between conservation and development parties, whilst some criticise it as oversimplification of complex natural wetland ecosystems. Of critical importance is whether both the functions and structure of a natural wetland can be replicated and recreated. To take Hong Kong as a case study, this chapter aims to critically discuss: (i) the need and state of wetland creation/restoration; (ii) the approach to, and characteristics of wetland mitigation; and (iii) the specific challenges of wetland recreation and restoration.

24.1. The Need to Create/Restore Wetlands

Wetlands were once regarded as wastelands, and even public health hazards for being a source of mosquitoes, flies, snakes, unpleasant odours, and disease. In the past, many thought that wetlands were places to avoid, or to drain and fill. Following an important proclamation of the Ramsar site of Mai Po and Inner Deep Bay in 1995 which signified an official recognition of wetlands by the Hong Kong SAR Government, the importance of wetlands is gaining increased recognition. People now acknowledge that wetlands are vanishing too rapidly, that they are important and that those remaining should be preserved, and those that have been destroyed or damaged should be restored and recreated. Wetlands in Hong Kong are mostly threatened by development, pollution and overuse

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(Lau, 2002). By far the greatest threat is environmentally irresponsible development.

Wetland creation and restoration is intended to help resolve contentious situations where development pressures conflict with wetland conservation efforts. This concept is gradually gaining acceptance and support, which is indicated by the growing number of wetland projects in Hong Kong. The decision to recreate or restore wetlands would bring with it a wide range of benefits, including provision of habitat for rare species and for rich biodiversity, flood abatement, soil conservation and pollutant removal from water, and an array of recreational values. Creation of wetland habitats is increasingly being used as an essential tool by conservationists in many parts of the world who wish to reverse the trend of habitat loss. However, it is of increasing concern that wetland creation in Hong Kong will be used as a tool by developers to push land development further into remote countryside, mostly in the New Territories.

24.2. Some of the Last Remaining Wetland Paradise in Hong Kong

Ironically, wetlands are often located where someone wants to build (e.g. housing development of Fairview Park, Royal Palms and Palm Springs). The New Territories can claim to be amongst the last remaining sources of wetlands in Hong Kong (Fig. 1). In the northwest, they are at Tin Shui Wai, Nam Sang Wai, Lut Chau, Tam Kon Chau, Mai Po, Pak Hok Chau, Lin Barn Tsuen, Wing Kei Tusen, Fung Lok Wai, Lok Ma Chau, Ma Tso Lung, San Tin and Long Valley. In the northeast, they are at Luk Keng, Nam Chung, Kuk Po, Sam A Chung, Sam A Tsuen and Sha Lo Tung. Some other wetlands can be found in the outlying islands, such as Pui O and Penny’s Bay in Lantau Island.

24.3. Some Current Wetland Mitigation Projects in Hong Kong

In the past few years, there is a growing trend for wetlands to be built alongside developments in Hong Kong (e.g. Hong Kong Wetland Park; West Rail Wetland Creation; Kam Tin Bypass; Lok Ma Chau Rail Station). On-site or off-site wetlands have been enhanced so that the total wetland value is increased. Figure 2 shows the locations of current wetland projects in Hong Kong. Table 1 shows some of the key ongoing wetland mitigation projects in Hong Kong.

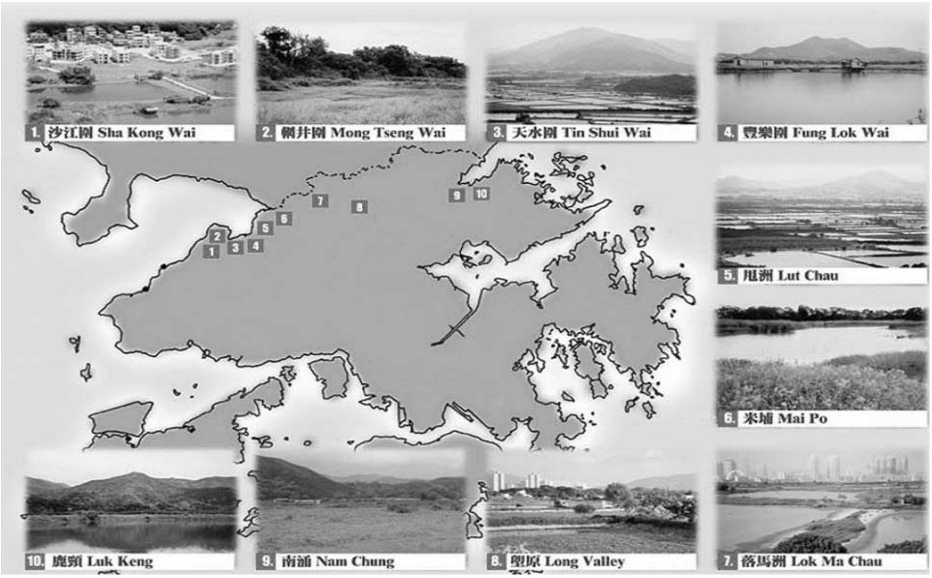

Figure 1: Some wetlands in the New Territories, Hong Kong.

24.4. Wetland Mitigation Approach

Wetland mitigation aims to offset the loss of wetland and wetland functions. More specifically, wetland mitigation aims to rectify the adverse impacts on the affected

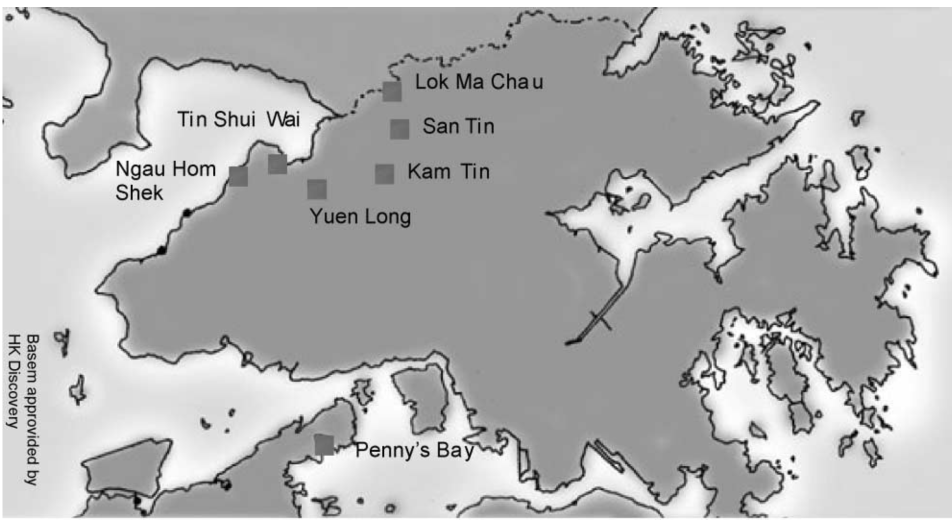

Figure 2: Location of current wetland projects in Hong Kong.

Table 1: Wetland projects in Hong Kong.

Location of the wetland	**Project**	**Method**	**Development purpose**	**Main use of the wetland**
Tin Shui Wai	Hong Kong Wetland Park	Creation and restoration	Urban development	Education, sightseeing and conservation
Lok Ma Chau	Lok Ma Chau Station Wetland Creation	Creation and enhancement	Transport	Conservation and water quality polishing
Lok Ma Chau	Sheung Shui to Lok Ma Chau Spur Line	Creation and enhancement	Transport	Conservation and fishery
Kam Tin	West Rail Wetland Creation	Creation and restoration	Transport	Conservation
Kam Tin	Kam Tin Bypass	Creation and restoration	Transport	Conservation
San Tin	San Tin Created Wetland	Creation and enhancement	Flood protection	Flood control and conservation
Tai Tam	Rare/Protected Species Transplantation, Hong Kong Disneyland	Creation	Theme park	Conservation
Yuen Long	Yuen Long Floodway Wetland Creation	Creation	Drainage	Water quality polishing, flood control

wetland environment by *restoration, enhancement or creation* (Kusler & Kentula, 1990). It is widely recognised that wetlands have important environmental, social, cultural and economic benefits, and some of the benefits are irreplaceable. Wherever possible, development should be avoided if it is to have significant impact on wetlands in Hong Kong. Where development is permitted due to overriding social or other benefits, impact should be minimised; where the impact is more extensive, wetland mitigation is required. Targeted mitigation works, with clear objectives carried out by experienced ecologists and contractors, can help to offset wetland loss thus providing an interface between expanding economy and development, whilst maintaining wetland values and functions. It is hoped that dramatic and controversial issues, such as Long Valley and Sha Lo Tung, can be avoided in the future.

In many countries (e.g. the US and the UK), wetlands have been created, restored or enhanced to encourage habitat replacement for years (Merritt, 1994). For each wetland that is impacted by a development project, the proponent of the project (be it a private company or the government) must restore the affected wetland after the project has been completed. If the wetland has been destroyed, then they are required to enhance the value of another existing wetland or create a new one (Garbisch, 1986). The goal in wetland mitigation is to attain a no net loss of wetland functions and values.

24.5. Suggested Mitigation Sequence

In all wetland projects, a three-step mitigation process, i.e. Avoidance → Minimisation → Compensation, should be used (Kusler & Kentula, 1990).

1. *Avoidance* — preservation of existing high quality habitat should take precedence over restoration or creation. Avoidance of impact is the first priority in any development project and alternative site or design for the project should be sought to prevent any wetland loss. It must be stressed that trading existing critical habitat for wetland restoration or creation should always be avoided unless the benefits of the trade-off outweigh the benefits of maintaining the critical existing habitat.
2. *Minimisation* — when there is no alternative to a development, adverse effects on wetland should be minimised.
3. *Compensation* — compensatory mitigation is required, as a last resort, for any unresolved impacts which remain after all appropriate and practical minimisation has been implemented. The emphasis is "no net loss" (National Research Council, 2001). Compensation should focus upon replacing specific wetland functions that have been lost as a result of the development. Both wetland functions and area are important.

Through mitigation measures, critical wetland habitats can be preserved. It is easier to restore a natural wetland than to create a brand new one. The success of mitigation banking is dependent on numerous variables (Hruby & Brower, 1994). It can take 10 years for a wetland to become established, however, it may take only several (e.g. three–five) growing seasons for a wetland to become established in subtropical Hong Kong as it has adequate rainfall and a long growing season. The fact that a mitigation bank can be created before the impact occurs allows time for the new or enhanced area to become effective, thereby eliminating any time without a wetland.

When a mitigation wetland site is chosen and planned, its location should be close to the impacted area (Kusler & Kentula, 1990). If an off-site location must be selected away from the impacted area, it should be located within the same hydrographic region. The reason for getting closer to the impacted site is that certain species, which are local or regional, is likely to be lost if the mitigated site is outside that particular area or region and thus outside their ecological range and tolerance. If a species has particular requirements with reference to geographical variation, the species would be susceptible to elimination. For example, if a species has a diet of very specific plants that can grow only at certain altitudes; the species would have limited or no areas to go to and could eventually die off, causing a loss in species richness and subsequently biodiversity for that area/region (Kusler & Kentula, 1990).

As a rule, wetland mitigation should not be carried out at the expense of high quality upland habitats which may have high wildlife or other values, but lack protection (Erwin, 1990).

"Type for type" approach in wetland habitat replacement should always be considered (Erwin, 1990; USACE, 1998). This approach tends towards restoration of a specific wetland habitat including soils, plants and wildlife. However, to advocate complete restoration of any type of wetland system in minute detail, is both impossible and unrealistic. This approach will not be appropriate when such replacement is not technically feasible, or when another type of wetland has greater value or more regional significance.

24.6. Wetland Replication

It is worth mentioning that wetland mitigation is NOT wetland replication as many may argue (Kusler & Kentula, 1990). No two wetlands are exactly the same. Unlike the cloned sheep Dolly, we cannot "clone" a wetland. In fact, wetland mitigation is all about conversion of a wetland from a disturbed or altered condition to a previously existing or similar condition.

The experiences from many wetland restoration projects in the US and the UK show that it is impossible to restore wetland ecosystems exactly as they were in terms of their complexity and variation in physical appearance, species composition and ecological processes (Keddy, 2000). This can never be achieved 100%, partly because wetlands are dynamic ecosystems, and partly due to lack of pre-impact baseline data. Most importantly, for economic, political, technical or practical reasons, compromises in the design/implementation of many wetland projects have to be made.

24.7. Challenges in Wetland Mitigation in Hong Kong

The experience of wetland creation and restoration to date in Hong Kong indicates that there are some challenges ahead, including:

24.7.1. Lack of Relevant Policy

Whilst there is a will to mitigate the effects of developments on wetlands, the capability is not always present. In Hong Kong, there is no standardised, consistent approach to wetland mitigation.

24.7.2. Lack of Clear Mitigation Goals and Objectives

There are many functions and values of wetlands, and these have to be clearly recognised before they can be restored or re-created under a mitigation scheme. Without clearly stated goals and objectives, projects lack direction (Erwin, 1990). Many mitigation projects lack clear goals and objectives so that crucial factors in their design could be missed, and the success of the mitigation may never be assessed. If the goal of a wetland project is quality enhancement of wildlife habitat, then the main objective is to improve habitat value for wetland birds. The failure to identify goals may suggest that the majority of the wetland projects were not designed by competent landscape architects. Most landscape architects have very little training in wetland ecology especially in specifying what type of wetland systems would be suitable for any particular form of wildlife, such as waterfowl. In reality, a range of experts in hydrological and civil engineering, soil science and landscape architecture, as well as constructed wetland specialists and wetland plant ecologists should be brought together for any wetland project. Of critical importance is the need to specify both immediate (e.g. temporary buffer area) and ultimate (e.g. wildlife habitat, education, research) goals.

24.7.3. Lack of Comprehensive Monitoring Methods and Requirements

In wetland creation/restoration projects, whether a project is a success or not is always the centre of argument between developers and conservationists. With clearly defined goals and objectives, associated performance standards should also be clearly stated (Erwin, 1990). An example can be the number of breeding pairs of several key wetland bird species (e.g. Great Egrets, Little Egret) that are to use the site after the completion of creation/restoration works. Quite often there is insufficient or complete absence of monitoring requirement for wetland projects. In most cases, monitoring methodology, which should be used for assessing the performance standards, is poorly defined. For some wetland projects, the monitoring methodology is inappropriate and unsatisfactory.

24.7.4. Lack of Reference Wetlands

For wetland creation/restoration projects, reference wetlands are used as a standard for measuring wetland ecosystem functions, such as enhancing value of wildlife habitat (see Brown, 1991). Wildlife populations are sampled and compared with those of reference wetlands. It is very difficult to locate a reference wetland which has suffered relatively little human disturbance in Hong Kong. Most of our wetlands have been impaired in one way or another by human influence, mostly in the form of direct impact from habitat development by either filling or related activity due to development pressure, or by altering the hydrology by traditional wet-farming activities over the past few decades, thus preventing them from acting as quality reference wetlands.

24.7.5. Lack of Appropriate Success Criteria

Success criteria, if any, may be predetermined and chosen inappropriately by engineers and/or landscape architects. A natural wetland ecosystem functions in a dynamic way, including a combination of biological, chemical and physical variables. Simple criteria may be chosen to signify success of a wetland project. If a wetland is created for the purpose of holding water and supporting wetland birds, the evaluation of success is simple and straightforward, and it is more likely to be a successful project. If we aim to recreate all the functions of a natural wetland then the project will almost certainly fail. Success criteria must be realistic, achievable, and defined carefully and appropriately with the involvement of qualified wetland specialists.

24.7.6. Lack of Basic Ecological Knowledge Regarding Natural Wetlands in Hong Kong

Hong Kong has very few, if any wetlands that have not been impacted by man. We know relatively little about aquatic plant communities native to the region, or how they are sustained. More ecological information is required about wetlands in Hong Kong or south China to establish the requirements of wetland plant and animal species so that these can be considered when designing new wetlands.

24.7.7. Lack of Local Qualified Expertise

The greatest obstacle to wetland mitigation and conservation in Hong Kong is the severe shortage of qualified local wetland specialists. There is no "Cook Book" approach for wetland restoration or creation (Kentula, 2003). Building wetland is not rocket science, but successful wetland mitigation depends greatly upon expertise in planning, project design and careful on-site expert supervision. Inappropriate design criteria are often associated with the failure of wetland projects. It is worrying that the majority of environmental consultants who play the role of "Constructed Wetland Specialist" for wetland projects in Hong Kong have very little training and experience in wetland ecology, creation and restoration. It is of increasing concern that some wetlands which are being created or restored (e.g. Wetland Creation Project in Kam Tin, Ming Pao, 2002) may not fully compensate for the loss of those which they are supposed to replace.

24.7.8. Inappropriate Scale and Location

Individual small creation or restoration projects targeting a single piece of wetland can be ecologically valuable. However, in some cases individual wetland patches are too small and they are isolated from one another, leading to drastic reduction of the potential ecological value being created (Kusler & Kentula, 1990). To avoid missing out connectivity, a rule of thumb is that whenever possible the minimum acceptable scale for wetland creation/restoration planning should be in a watershed context. Created or restored wetlands are more likely to be successful if their functions are considered in terms of their role in the overall surrounding watershed. Wetlands which are located at a hydrologically and ecologically appropriate place may favour the establishment and maintenance of wetland functions and values. In addition, wetland projects should incorporate upland habitats to form an inter-connected ecosystem.

24.7.9. Lack of Long-Term Commitment for Created/Restored Wetlands

A matter of great concern is the apparent lack of a follow-up plan in connection with the created/restored wetland. Where successful wetland creation is recognised, it is questionable as to whether long-term continuous management, which is costly would be considered, undertaken and strictly enforced. Water quality and level have to be monitored and regulated. Vegetation management is also a must to avoid the dominance of a few species and to control the invasion of unwanted exotic, aggressive species, such as *Phragmites* spp., *Echinochloa* spp. and *Brachiaria* spp.

24.7.10. Rigid Specification of Civil Engineering Projects

Most wetland creation projects in Hong Kong are considered as parts of the landscape section of civil engineering projects, not ecological. Strict contractual agreements have to be adhered to. In wetland mitigation, on-site flexibility in habitat construction is crucial, with habitat designers on site constantly being aware of opportunities for enhancements in ground profiling, surface treatments and planting layouts which go beyond the base plans. A wetland creation project cannot be carried out using traditional construction methods and contracts. However, the great skills amongst our construction industry workers can be harnessed and applied to the task with good effect if they are properly instructed by experienced wetland creation staff and ecologists, and with a new type of contractual arrangement. Bull-dozer operators, for example, often need guidance with regard to critical elevation requirements and micro-profiling of the ground surface. Labourers need guidance with regard to plant handling and planting requirements.

24.8. Conclusion

Despite the fact that a number of wetland projects are underway, in technical terms, wetland creation and restoration is still in its infancy in Hong Kong. Wetland mitigation remains a challenge to conservationists and wetland specialists involved in creating, restoring or enhancing dynamic, complex wetland ecosystems, which would otherwise be formed by natural forces. There is plenty of room for improvement of the wetland mitigation approach and technique. Extra scientific and technical efforts should be put into improving wetland mitigation, covering all phases of project planning, design, construction, and operation and maintenance.

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Glossary[1]

Wetland restoration	to return a wetland from a disturbed condition to a previously existing or similar condition.
Wetland enhancement	to increase some functions of an existing wetland.
Wetland creation	to build new wetlands by converting non-wetlands into wetlands.
Wetland functions	the driving forces which maintain wetland ecosystems, e.g. hydrology, physical processes (e.g. sediment movement), biological processes (e.g. competition, predation), and biogeochemical processes (e.g. nutrient cycling).
Wetland values	such as use value (e.g. recreation), social value (e.g. water quality improvement, flood protection), wildlife value (e.g. breeding, rearing and feeding ground).
A hydrographic region	is defined as a geographic area drained by a major stream, or an area composed of a drainage system made up of streams and lakes.
Goals	are defined as general statements about desired project outcomes.
Objectives	are defined as specific statements about desired project outcomes.
Performance standards	are observable or measurable attributes that can be used to determine if a project meets its intended objectives.

[1] See Kusler & Kentula (1990) and Merritt (1994).

Subject Index