Hannah S. Elliot
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Editors

River Ecosystems

Dynamics, Management and Conservation

Earth Sciences in the 21st Century

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RIVER ECOSYSTEMS: DYNAMICS, MANAGEMENT AND CONSERVATION

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RIVER ECOSYSTEMS: DYNAMICS, MANAGEMENT AND CONSERVATION

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PREFACE

Management and restoration of river ecosystems are based on an understanding of the relations between physical, chemical and biological processes at varying time scales. This new book examines river ecology, disturbances of the ecology and restoration strategies. Topics discussed include the social side of river management; historical change and management of the floodplain forests of the Middle Elbe River; pollution, water quality, toxic contamination and fish health of the Volga River and the relationship between river management and economic growth in urban regions.

Chapter 1 - Management and restoration of river ecosystems are based on an understanding of the relations between physical, chemical, and biological processes at varying time scales. Often, human activities have accelerated the temporal progression of these processes, resulting in unstable flow patterns and altered biological structure and function of stream corridors. This chapter discusses river ecology, disturbances of the ecology, and restoration strategies.

Chapter 2 - Freshwater ecosystems play a primary role in the biosphere, which justifies interest in the assessment their health or condition. One important component of the biological assessment of stream conditions is evaluating the direct or indirect effects of human activities or disturbances. Most biological assessments are based on the concept of comparing current conditions to natural conditions in the absence of human disturbance.

In Europe, increasing water quantity and quality problems have led to the development of an integrated approach for water management systems, including all water-related impacts, which resulted in the Water Framework Directive (WFD) in 2000. The Reference Condition Approach concept has been adopted by the WFD as it requires the evaluation of the ecological status and this may be expressed as a deviation from the near natural reference condition. According to the WFD, reference conditions should be linked to stream typology, and reference sites should present the full range of conditions expected to occur naturally within a given stream type.

In the Mediterranean streams context, a methodology to select reference sites based on the application of twenty a priori criteria that reflect the characteristics of Mediterranean streams and their most frequent disturbances, has been a critical step for the establishment of Mediterranean reference conditions. Subsequently, the development of a typology of Mediterranean streams resulted in five ecotypes (four permanents and one temporary), showed that stream classification schemes based on environmental variables need to be validated by biological variables.

The WFD establishes that ecological status must be classified into five quality classes (high, good, moderate, poor and bad) by means of biological indicators. In this context, a study that analysed the sensitivity of diverse macroinvertebrate metrics to a multiple stressor gradient, which includes the pressures in a Mediterranean area, revealed that the IBMWP and ICM-11indices ensure a proper classification of the ecological status of Mediterranean streams. However, much less is known about the physico-chemical quality of these streams.

An important conclusion drawn from studies conducted in Mediterranean streams indicates the difficulty in assessing temporary streams as regards both biological and physicochemical quality. The diverse macroinvertebrate communities within this type of streams, mainly due to the wide variability in hydrology conditions, could justify further division into subtypes. In relation to physico-chemical quality, temporal streams show an important spatial and temporal variability in solute concentrations. Both low surface flow and spatial intermittency reinforce the effects of differences in advection, water residence time, biological community structure, sediment-water interactions or environment redox conditions through stream channels. By way of example, all these factors influence the rates of the biogeochemical processes involved in the nutrients cycle. Consequently, biogeochemical process rates and reactions will have a patchy distribution in temporary streams. At the same time, this local stream-channel environmental conditions change over time as flow discharge changes, which increases the variability of physicochemical quality on the temporal scale.

Chapter 3 - The present volume contains eight chapters on river management. The background of most of them is formed by river flood risk, which is usually assumed to increase in the near future due to climate change. ‘Room for the Rivers’ is the general catchword for the new policies in Europe that aim to move away from harnessing the rivers between ever-higher dikes and instead give the rivers more horizontal space to drain their waters.

Room-for-river policies will intensify the social aspects of river management in two ways – one negative, one positive. On the negative side, giving more space to rivers will often imply that land will have to be given back to rivers or that land use will have to change: houses in floodplains will have to be removed or be made flood-proof, famers may have to extensify their operations and confront lower land values, and recreationists may find their forest ‘ecologically flooded’ when they arrive with their picnic baskets.

On the positive side in the majority of cases, giving space back to rivers results in a re-naturalization of the riverine landscape and a restoration of ecological process and species diversity. This as such is already a great benefit for conservation, recreation and quality of life in riparian communities.

Additionally, this natural quality improvement may also be sold to outsiders such as urban tourists and house seekers, generating local economic benefits. And finally, room-for-river policies offer a window of opportunity to riparian villages, towns and cities to re-connect to their rivers in new designs of river fronts and river-based activities.

On the local level, this pattern of both negative and positive potential impacts of the new river policies may give rise to integrated room-for-river packages in which social costs and social benefits are mixed into net win-win designs for both river managers and communities. This has been the source of inspiration for the EU-funded ‘Freude am Fluss’ project on which much of the material presented in this special issue has been based.

Chapter 4 - Floodplain forests are vital to the functioning of river ecosystems. They support the rich diversity characteristic of these systems, play an important role in nutrient cycling, and serve as an ecotone between the terrestrial and aquatic biomes. This chapter explores the changes that have taken place in the floodplain forest community of the Middle Elbe River over the last 50 years, examining the role that altered river dynamics, herbivore pressure, human management, and invasive species have played in shaping the current forests. It also looks at the development of the UNESCO Flusslandschaft Elbe Biosphere Reserve, which now manages much of the Middle Elbe's ecosystem, and the influence the biosphere reserve's activities have on the forests. The authors' research indicates that the floodplain forests, currently dominated by pedunculate oak (*Quercus robur)* and field maple (*Acer campestre*), have undergone a shift in community composition since the 1950's, including a loss of more flood tolerant species. The authors' results further suggest that browsing is limiting regeneration. When examining the forests across the different management zones, it appears that the core zone forests will likely experience an eventual loss of oak dominance, whereas forestry practices in the buffer zones should maintain the oak at a moderate level. American green ash (*Fraxinus pennsylvanica*), an introduced species, is also more prevalent in the buffer zones. The different zones of the biosphere reserve currently help maintain a more diverse mix of forest communities across the landscape. However, without a restoration of a more natural flooding regime, early successional communities will become increasingly rare. The biosphere reserve managers recognize this issue and are currently pursuing several projects to help mitigate the loss of natural processes, including setting back levees to increase the size of the active floodplain, restoring backwater lakes, and removing introduced species.

Chapter 5 - In river management as in most other fields of environmental planning, "all stakeholders" are supposed to somehow participate. But who will actually sit in, who will actually have a voice? This question is of great consequence in practice, but underdeveloped in literature.

Taking most of its examples from the field of river management, the present paper aims to shed some light on the dark road that leads from "all stakeholders" to a balanced representation of stakes in participatory, collaborative planning.

Their method is to develop the initially undifferentiated cloud of stakeholders through five steps: (1) from all stakeholders to significant stakeholders, (2) framing reflection, (3) representation of significant stakeholders, (4) setting a balanced scene, and (5) maintaining a balanced process.

The authors then attach a number of categories, examples, reflections and sometimes a few rules to each of these steps. This system serves as an informal ethics to improve the quality and justifiability of who is to represent what in participatory planning.

Chapter 6 - The characteristic of current state of Volga river is given. Concentrations of organic and inorganic toxic substances in water are reported. Basic clinical and postmortem signs of fish intoxication are described; changes in the cellular structure of their organs and tissues, as well as disturbances in hemogenesis, developing under the effect of toxic agents, are characterized. The main disturbances to fish caused by the accumulation of microelements in their organs and tissues are also considered. Based on dose–effect dependencies calculated with respect to the total concentration of toxic substances, standardized to MPC, and fish health criteria, cases that exceed the critical levels of pollutants are demonstrated for the investigated river sections

Chapter 7 - The Serengeti ecosystem is often taken to be the 25000 km^2 animal migration area (Figure 1a). This includes the 14,763 km^2 Serengeti National Park (SNP), the Masai Mara Reserve in Kenya, and a number of game controlled areas that form a buffer zones, principally the Maswa, Ngorongoro, Loliondo, Ikorongo, Grumeti, and the Speke Gulf Game Controlled Area (SGGCA) that, although tiny (95 km^2), is potentially important because, if human encroachment was removed, it would provide access for wildlife to the permanent waters of Lake Victoria (Figure 1b). However this definition of the ecosystem ignores the hydrology. The Serengeti ecosystem has only one perennial river, the Mara River. The Mara River, together with a few scattered springs in the northern region of the SNP, is the only source of water for migrating wildlife in the dry season in a drought year. Thus the source of Mara River water in the dry season, namely the Mau forest in Kenya's highlands, is also part of the Serengeti ecosystem even if the migrating animals do not migrate to that area (Gereta et al., 2002 and 2009).

Lake Victoria (Figure 1c) has a surface area of 96,000 km^2. Its shores are shared by Uganda, Kenya, and Tanzania. It drains 25 large rivers with a total catchment area of about 184,000 km^2, supporting 30 million people. The lake is large enough (see below) to affect rainfall over the Serengeti.

Chapter 8 - Temporary streams represent a significant yet understudied and particularly vulnerable portion of river networks. While the vast majority of stream and river research to date has focused on perennial flowing waters, recent work reveals that temporary streams are not only abundant and widely distributed, but also play a significant role in the hydrological and ecological integrity of lotic networks. In this chapter, we seek to summarize the current state of the science of these ubiquitous portions of river networks while simultaneously stressing the need for their future investigation. The authors begin by defining temporary streams and their hydrology and highlighting their abundance and extent. We then consider the ecological significance of temporary streams, including their role as faunal and floral habitat providers, biogeochemical processors, and connectivity corridors within river networks. The chapter concludes with a discussion of policy issues surrounding temporary streams and the anthropogenic disturbances they face.

Chapter 9 - Sulfuric acid is discharged from acid sulfate soils or mining sites in the watershed of rivers, and this sulfuric acid affects aquatic communities. In basins with coal mining sites, sulfate migrates into river systems via mine drainage as well as via ground water flow.

In order to evaluate the non-point contamination of sulfate via ground water into river ecosystems, the authors analyzed the chemical properties of river water from the upper-most basins to the river mouth at 1-2 km intervals.

The authors surveyed two rivers in the northern part of Kyushu Island, Japan. The Ongagawa River basin has many abandoned coal mining sites and the SO_4^{2-} /Cl^- concentration ratio (w/w) reached up to 7.0 in the middle basin, whereas the ratio in the Chikugogawa River with few coal mining sites in the basin was constantly 1.0-2.0 in the middle basin, in contrast to the extremely high value (20.0) at the upper-most basin of the river because of the effect of sulfur emissions from active volcanoes. Thus, we found that the SO_4^{2-} /Cl^- concentration ratio of river water with abandoned coal mining sites in its basin was much higher than was the case with a river without coal mining sites in the basin. The SO_4^{2-} /Cl^- concentration ratio reached a higher value in the basin of a tropical peat swamp forest with acid sulfate soils in central Kalimantan, Indonesia, and the ratio can be used as a useful index for the qualitative evaluation of the non-point contamination of sulfates in river systems.

Chapter 10 - Water stress in northern China has intensified water use conflicts between upstream and downstream areas and also between agriculture and municipal/industrial sectors year by year. In this study, the NIES Integrated Catchment-based Eco-hydrology (NICE) model (Nakayama, 2008a, 2008b, 2010a, 2010b; Nakayama and Fujita, 2010; Nakayama and Watanabe, 2004, 2006, 2008a, 2008b; Nakayama *et al*., 2006, 2007, 2010) was applied to the Biliu River catchment, northern China, to estimate the carrying capacity of the water resource there. The model reasonably backcasted the degradation of water resources such as river discharge and groundwater after the completion of the reservoir in the middle reach of the river. The normalized difference vegetation index (NDVI) calculated from NOAA/AVHRR satellite image clearly showed vegetation degradation downstream of the reservoir. Furthermore, statistical analysis of a decoupling indicator (OECD, 2001) based on the water carrying capacity simulated by NICE and on the satellite data of vegetation index indicated that water-related stress in Dalian city, where the economy has grown rapidly after the completement of the reservoir, has increased in accordance with the environmental degradation below the reservoir. These results indicate a close relationship between water resource and economic growth, which has greatly affected ecosystem degradation and its serious burden on the environment in the catchment. The simulated results also highlight the linkage between urban development in Dalian and sustainable water resource management, and this assessment of the interactions between the sites of water source and demand would support decision-making on sustainable development in the catchment.

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Chapter 1

RIVER ECOLOGY AND STREAM RESTORATION

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Management and restoration of river ecosystems are based on an understanding of the relations between physical, chemical, and biological processes at varying time scales. Often, human activities have accelerated the temporal progression of these processes, resulting in unstable flow patterns and altered biological structure and function of stream corridors. This chapter discusses river ecology, disturbances of the ecology, and restoration strategies.

1. RIVER ECOSYSTEMS

1.1. Spatial Elements of River Ecosystems

Ecosystems of rivers vary greatly in size. Taking a deeper look into these ecosystems can help to explain the functions of landscapes, watersheds, floodplains and streams, as shown in Figure1. In ecosystems movement between internal and external environments is common. This may involve movement of materials (e.g. sediment and storm water runoff), organisms (e.g. mammals, fish and insects) and also energy (e.g. heating and cooling of stream waters).

Many sub-ecosystems form a river ecosystem which, in turn, can also be part of a larger scale landscape ecosystem. The structure and functions of the landscape ecosystem are in part determined by the structure and functions of the river ecosystem. The river ecosystem may have input or output relations with the landscape ecosystem, thus, the two are related. In order to plan and design a river ecosystem restoration, it is vital to first investigate the relations between the ecosystems. Landscape ecologists use four basic terms to define spatial structure at a particular scale:

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1. Matrix - the land cover that is dominant and interconnected over the majority of the land surface. Theoretically the matrix can be any land cover type but often is it forest or agriculture.
2. Patch - a nonlinear area (polygon) that is less abundant than, and different from, the matrix.
3. Corridor - a special type of patch that links other patches in the matrix. Usually, a corridor is linear or elongated in shape, such as a stream corridor.
4. Mosaic - a collection of patches, none of which are dominant enough to be interconnected throughout the landscape.

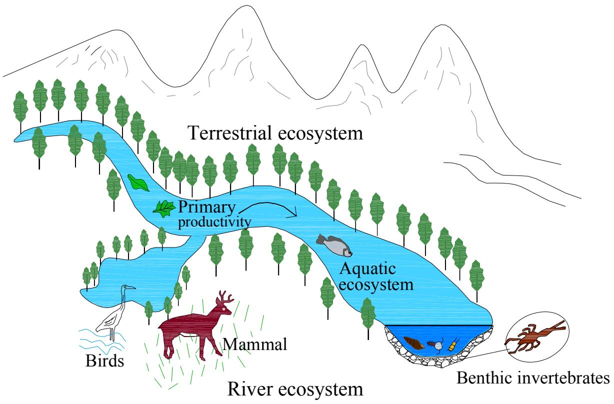

Figure 1. A river ecosystem consists of the terrestrial ecosystem and the aquatic ecosystem, which is affected by and impacts on the landscape ecosystem through input and output (after FISRWG, 1997).

(a) (b) (c) (d)

Figure 2. (a) Forest matrix in the suburbs of Beijing, China; (b) A township patch and a surrounding stream corridor in Wasserburg, Germany; (c) Stream corridor (the Leinbach River in Germany) and riparian forest matrix; and (d) Mosaic consisting of forest, lake, island, and hills (Banff, Canada).

Figure 2. shows examples of a forest matrix, a city patch, a stream corridor, and a mosaic consisting of a lake, island, forest and hills. One may see a matrix of mature forest, cropland, pasture, clear-cuts, lakes, and wetlands on a landscape scale. However, on a river reach scale, in a matrix of less desirable shallow waters, a trout may perceive pools and well sheltered, cool, pockets of water as preferred patches and in order to travel safely among these habitat patches, the stream channel may be its only alternative. The matrix-patch-corridor-mosaic model is a very useful, basic way of describing structure in the environment at all levels. When planning and designing ecosystem restoration, it is very important to always consider multiple scales.

The stream corridor is an ecosystem with an internal and external environment (its surrounding landscape). Stream corridors often serve as a primary pathway for the aforementioned movement of energy, materials, and organisms in, through, and out of the system. This may be accomplished by connecting patches and functioning as a conduit between ecosystems and their external environment. Movement in, through, and out of the ecosystem may be dictated by spatial structure, especially in corridors; conversely, this movement also serves to change the structure over time. Thus, the end result of past movement is the spatial structure, as it appears at any point in time. In order to work with ecosystems at any scale it is paramount to understand the feedback loop between movement and structure.

Many of the functions of the stream corridor are strongly interlinked with drainage patterns. So, many people commonly use the term 'watershed scale', and it will also be used in this chapter. A *watershed* is defined as an area of land that drains water, sediment, and dissolved materials to a common outlet at some point along a stream channel (Dunne and Leopold, 1978). Watersheds, therefore, occur at many different scales, ranging from the watersheds of very small streams that measure only a few km^2 in size to the largest river basins, such as the Yangtze River watershed. The matrix, patch, corridor, and mosaic terms can still be used to describe the ecological structure within watersheds. However, one could further describe the watershed structure more meaningfully by also focusing on elements such as upper, middle, and lower watershed zones, drainage divides, upper and lower hill slopes; terraces; floodplains; estuaries and lagoons; and river mouths and deltas. Figure 3. displays examples of (a) the upper watershed (the Yangtze River at the Shennongjia Mountain); (b) a mountain stream (the Qingjiang River is a tributary of the Weihe River in the Yellow River Basin); (c) an alluvial river (the Blue Nile at the confluence with the White Nile River); and (d) an estuary (the Venice Lagoon at the Po River mouth).

The river corridor is a spatial element (a corridor) at the watershed and landscape scales. Common matrices in stream corridors include riparian forest or shrub cover or alternatively herbaceous vegetation. Examples of patches at the stream corridor scale are wetlands, forest, shrub land, grassland patches, oxbow lakes, residential or commercial development, islands in the channel, and passive recreation areas such as picnic grounds.

Figure 4. shows a cross section of a river corridor. The river corridor can be subdivided by structural features and plant communities. Riparian areas have one or both of the following characteristics: (1) vegetative species clearly different from nearby areas; and (2) species similar to adjacent areas but exhibiting more vigorous or robust growth forms. Riparian areas are usually transitional between wetland and upland

Figure 3. (a) Upper watershed of the Yangtze River at the Shennongjia Mountain; (b) the Qingjiang River in the Yellow River Basin; (c) the Blue Nile at the confluence with the White Nile in Sudan; and (d) the Venice Lagoon at the Po River mouth in Italy.

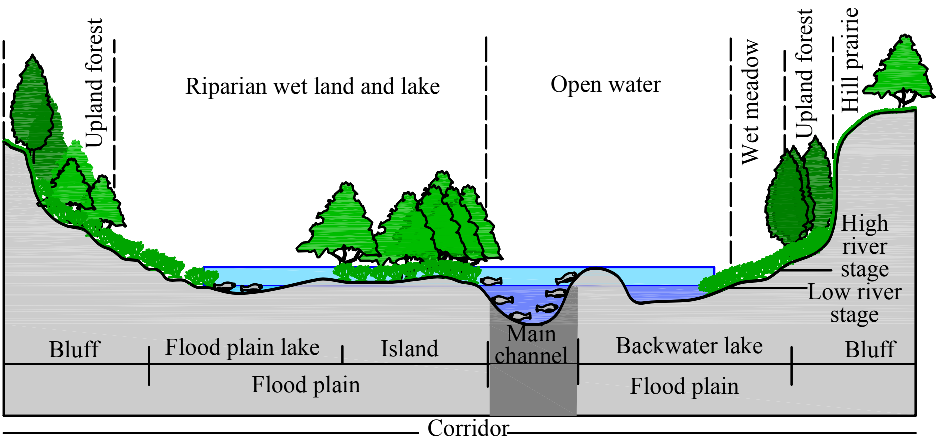

Figure 4. A cross section of a river corridor, in which the river corridor is subdivided by structural features and plant communities (after FIRSWG, 1997).

1.2. Biological Communities

The biological community of a river ecosystem is determined by the characteristics of both terrestrial and aquatic ecosystems. The terrestrial ecosystem depends mainly on the plant community and the aquatic ecosystem comprises aquatic plants, benthic invertebrates, and vertebrates (fish, reptiles, and amphibians).

1.2.1. Terrestrial Ecosystem

The terrestrial ecosystem comprises the plant community, amphibians, reptiles, mammals and birds. The plant community is an essential basic element for the stream ecology. The ecological integrity of a river is directly related to the integrity and ecological characteristics of the plant communities that make up and surround the river. The biological communities depend on these plant communities as a valuable source of energy. The plants also provide a physical habitat and moderate solar energy fluxes to and from the surrounding aquatic ecosystem. Figure 5. shows the terrestrial plant community by the upper Dadu River in Sichuan Province, China. The high density of vegetation develops in the valley and on the slopes, which provides the primary product for the aquatic ecosystem. The primary product for ecology is provided by plant communities. Only a small amount of this organic material is stored as above- and below-ground biomass, while senescence, fractionation, and leaching to the organic soil layer in the form of leaves, twigs and decaying roots are the components of the annual loss of a significant fraction of organic matter. This annual loss of organic matter represents a major storage and cycling pool of available carbon, nitrogen, phosphorus and other nutrients, for it is rich in biological activity of microbial flora and microfauna. The characteristics of the plant communities directly influence the diversity and integrity of the faunal communities. In general, complex floral community supports complex faunal communities. The faunal composition is a function of the following habitat features of river corridors: 1) permanent water; 2) high primary productivity and food availability; 3) spatial and temporal contrasts in cover types; 4) critical microclimates; 5) horizontal and vertical habitat diversity; 6) effective seasonal migration routes; 7) high connectivity between vegetated patches. Following disturbances to the land, whether they be naturally occurring or resulting from human activity, plant succession occurs, in which pioneer species well-adapted to bare soil and plentiful light are gradually replaced by longer-lived species that can regenerate under more shaded and protected conditions. The most common natural sources of disturbance within a river valley are flooding and channel migration.

Figure 5. Terrestrial pant community by the Dadu River in Sichuan Province, China. A high density of vegetation develops in the valley and on the slopes, which provides primary product for the aquatic faunal community.

Knowledge of natural succession in a stream corridor can be very useful for restoration practitioners and should be taken advantage of by planting hardy, pioneer species to stabilize an eroding bank, while planning for the eventual replacement of these species by longer-lived and higher-succession species.In arid and semi- arid areas, water is vital for the fauna since the only naturally occurring permanent sources of water present are the streams.

The high primary productivity and biomass of riparian areas is largely a result of these moist environments, for the food sources and cover types of the surrounding areas are very different from the riparian area. Stream corridors provide water, shade, evapotranspiration, and cover, thus, ameliorating the temperature and moisture extremes of uplands. Nearly all amphibians, many reptiles and mammals are found primarily in river corridors and riparian habitats.

1.2.2. Aquatic Ecosystem

Stream biota are often classified into seven groups—bacteria, algae, macrophytes (higher plants), protists (amoebas, flagellates, ciliates), micro-invertebrates (invertebrates less than 0.5 mm in length, such as rotifers, copepods, ostracods, and nematodes), macro-invertebrates (invertebrates greater than 0.5 mm in length, such as mayflies, stoneflies, caddisflies, crayfish, worms, clams, and snails), and vertebrates (fish, amphibians, reptiles, and mammals) as shown in Figure 6. Undisturbed streams can contain a remarkable number of species. For example, more than 1,300 species were found in a 2 km reach of a small German stream, the Breitenbach, when a comprehensive inventory of stream biota was taken. The most important elements of the aquatic ecosystem for river management are aquatic plants, benthic invertebrates, and vertebrates (fish, reptiles, and amphibians).

Aquatic plants usually consist of mosses attached to permanent stream substrates and macrophytes including floating plants, such as *Eichoimia crassips*, submerged plants, such as *Potamogeton*, and emergent plants, such as *Phragmites communis trirn* (reed). These plants provide primary productivity for the faunal community, and play an important role in decontaminating the river water and providing multiple habitats for fish and invertebrates. Bedrock or boulders and cobbles are often covered by mosses and algae. Figure 7. shows microhabitats with moss on cobbles, submerged macrophytes species *Potamogeton*, floating plants species *Lemna minor L.*, and emergent plants species *Phragmites communis trirn* (reed). Rooted aquatic vegetation may occur where substrates are suitable and high currents do not scour the stream bottom. Luxuriant vascular plants may occur in some areas where water clarity, stable substrates, high nutrients, and slow water velocities are present. Benthic invertebrates collectively facilitate the breakdown of organic material, such as leaf litter, that enters the stream from external sources.

Larger leaf litter is broken down into smaller particles by the feeding activities of invertebrates known as shredders (insect larvae and amphipods). Other invertebrates filter smaller organic material from the water, known as filterers (blackfly larvae, some mayfly nymphs, and some caddisfly larvae); scrape material off the surfaces of bedrock, boulders, and cobles, known as scrapers (snails, limpets, and some caddisfly and mayfly nymphs); or feed on material deposited on the substrate, known as collectors (dipteran larvae and some mayfly nymphs) (Moss, 1988). Some macro-invertebrates are predatory, known as predators, such as dragonfly, which prey on small vertebrates. Figure 8. shows typical species of the five groups with different ecological functions.

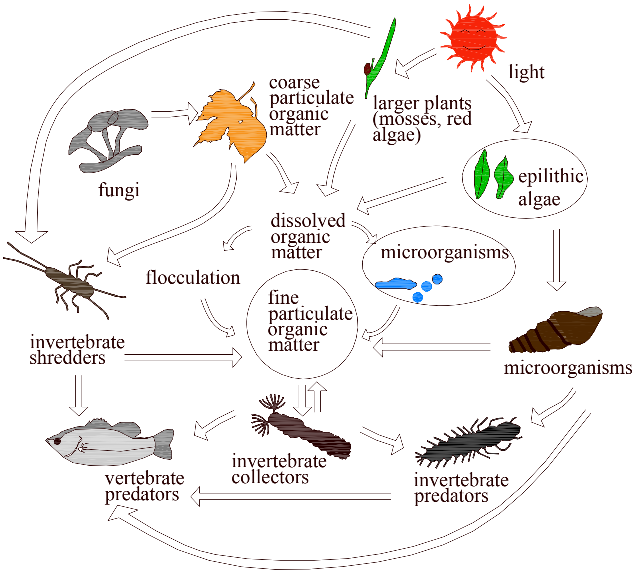

Figure 6. Stream eco-system and bio-community (after FISRWG, 1997).

Figure 7. Aquatic plants: (a) moss on cobbles; (b) Potamogeton; (c) Lemna minor L.; and (d) Phragmites communis trirn.

Figure 8. Typical species of the five groups of benthic invertebrates with different functions in the food chain: (a) Gomphidae (Dragonfly-predator); (b) Viviparidae (scraper); (c) Hydropsychidae (collector-filterer); (d) Corbiculidae (collector-filterer); and (e) Haliplidae (shredder).

Fish are the apex predator in the aquatic system. Many restoration projects aim at restoration of fish habitat. From the headwaters to the estuaries the composition of fish species varies considerably due to changes in many hydrologic and geomorphic factors which control temperature, dissolved oxygen, gradient, current velocity, and substrate. The amount of different habitats in a given stream section is determined by a combination of these factors. Fish species richness (diversity) tends to increase downstream as gradient decreases and stream size increases. For small headwater streams, the gradient tends to be very steep and the stream is small and environmental fluctuations occur with a greater intensity and frequency, therefore, species richness is lowest (Hynes, 1970).

Some fish species are migratory and travel long distances to return to a certain site to spawn. They have to swim against currents and go up over waterfalls, thus, showing great strength and endurance. When migrating they move between saltwater and freshwater, therefore, need to be able to osmoregulate efficiently (McKeown, 1984). According to their temperature requirements, species may also generally be referred to as cold water or warm water and gradations between. Salmonid fish prefer cold and highly oxygenated water, and, therefore, can generally be found at high altitudes or northern climes. Salmonid populations are very sensitive to change or deterioration of their habitat, including alteration of flows, temperature, and substrate quality. They tolerate only very small fluctuations in temperature and only reproduce under certain conditions. Their reproductive behavior and movements are affected by almost undetectable changes in temperature. Usually a salmonid spawns by depositing eggs over or between clean gravel, which remain oxygenated and silt free due to upwelling of currents between the interstitial spaces. Salmonid populations, therefore, are highly susceptible to many forms of habitat degradation, including alteration of flows, temperature, and substrate quality.

The general concern and interest in restoring habitats for fish by improving both quality and quantity, is due to the widespread decrease in numbers of native fish species. With ecological, economical, and recreational considerations in mind, the importance given to the restoration of fish communities is increasing. In 1996 approximately 35 million Americans went fishing for recreational purposes resulting in over $36 billion in expenditures (Brouha, 1997).

Since most recreational fishing is in streams, it is important to restore stream corridors. Restoration activities have often been focused on improving local habitats, such as fencing or removing livestock from streams, constructing fish passages, or installing instream physical habitat. However, the success of these activities, demonstrated by research, has been very small or questionable. Over its life span, a species needs many resources and has a great range of habitat requirements which were not considered during the restoration.

Although the public are most interested in fish stocks, another goal of the stream restoration is to preserve other aquatic biota. Of particular concern are freshwater mussels, many species of which are threatened and endangered. Mussels are highly sensitive to habitat disturbances. Some of the major threats faced by mussels are dams, which lead to direct habitat loss and fragmentation of the remaining habitats, persistent sedimentation, pesticides, and exotic species like fish and other mussel species which are introduced into the habitat.

1.2.3. Habitat

The diversity of available habitats directly influences the biological diversity and abundance in streams. When a stream system is stably functioning, the diversity and availability of habitats is promoted. This is one of the primary reasons stream stability is always considered in river ecosystem restoration activities.

Figure 9. (a) shows the gravel and cobbles bed of Juma River near Beijing, which is very stable even during flood events. The stable habitat is colonized by a biocommunity with high diversity . Figure 9. (b) shows the Dabai River in the upper Yangtze River basin, which channel bed has a very unstable due to high sediment load. There is few species in the Dabai River and the biodiversity is low.

Under less disturbed situations, a narrow, steep-walled cross section provides less physical area for habitat than a wider cross section with less steep sides, but may provide more biologically rich habitat in deep pools. Habitat increases with stream sinuosity. A uniform sand bed in a stream provides less potential habitat diversity than a bed with debris dams, boulder cascades, rapids, step-pool sequences, pool-riffle sequences, or other types of “structures”.

A river may be seen either as a single functioning entity or an ecosystem with components which interact and have numerous connections between them. In order to restore an ecosystem successfully, one must not forget these fundamental relations. Certain topographic settings may be more likely than others to be subject to periodic, dramatic changes in hydrology and related vegetation structure as a result of massive woody debris jams. The structural aspects of vegetation may affect the functions of the stream ecosystems. Vegetation and detritus influence the course of water in an undeveloped watershed. Balanced interactions between terrestrial and aquatic systems result from the mobilization and conservation of nutrients in complex patterns. Land uses such as agriculture and livestock grazing and the flow patterns of water, sediment, and nutrients cause the characteristics and distribution of vegetation to change.

Figure 9. (a) Gravel and cobbles bed of the Juma River near Beijing is stable, which is colonized by a biocommunity with high diversity; and (b) The Dabai River in the upper Yangtze River basin is very unstable due to high sediment load and has a very low biodiversity.

The most obvious roles of plants are those that affect fish and wildlife. At the landscape level, the fragmentation of native cover types has had a negative impact on the wildlife needing large areas of ongoing vegetation.

In some systems, small fractures in the continuity of corridors can greatly affect animal movement and can negatively impact the conditions in a stream which permit it to host certain aquatic species.

Narrow stream corridors that are essentially edge habitat may encourage generalist species, nest parasites and predators, and where corridors have been established across historic barriers to animal movement, they can disrupt the integrity of regional animal assemblages (Knopf et al., 1988).

The conditions of nearby habitats must be taken into account when planning to restore riparian areas. Carothers (1979) found that non-riparian birds frequently use the edges of riparian areas as habitats.

However, smaller riparian birds carry out activities within the riparian area during the breeding season and larger species often forage in nearby areas (Carothers et al., 1974). In fact although the larger species may forage many miles away from the riparian area, they would still depend on them critically (Lee et al., 1989).

If the conditions upstream of an ecosystem have been significantly altered by, for example, a dam or any other water diversion project, it will be impossible to restore that ecosystem to its natural, perfect condition. This stands even if complete restoration is a possibility.

For example, the creation of areas of woody vegetation in the channel below several dams in the Platte River Valley in Nebraska has significantly decreased areas of wet meadow – an important habitat. The area has been declared a critical habitat for the Whooping Crane, the Piping Plover, and the Interior Least Tern (Aronson and Ellis, 1979).

1.3. Ecological Conditions

Flow - Streams are distinguished from other ecosystems by a flow of water from upstream to downstream. The micro- and macro- distribution patterns of many stream species are affected by the spatial and temporal characteristics of stream flow, such as fast versus slow, deep versus shallow, turbulent versus laminar, and flooding versus low flows (Bayley and Li, 1992: Reynolds, 1992; Ward, 1992).

Flow velocity affects the deliverance of food and nutrients to organisms, however, it can also dislodge them and prevent them from remaining at a certain site. When a stream has a very slow flow, the fauna on the banks and the bed are similar in composition and configuration to those present in stagnant waters (Ruttner, 1963). High flows are cues for timing migration and spawning of some fish. When fish detect high flows, some will migrate and some will spawn.

Temperature - Water temperature can vary markedly in a stream system and between different stream systems. It is a very important factor for cold blooded aquatic organisms for it affects many of their physiological and biochemical processes.

Stream insects, for example, often grow and develop more rapidly in warmer portions of a stream or during warmer seasons. Some species may complete two or more generations per year at warmer sites yet only one or fewer at cooler sites (Sweeney, 1984; Ward, 1992). This can also be applied to algae and fish for their growth rates increase with increased water temperature (Hynes, 1970; Reynolds, 1992). Some species are only found in certain areas due to the correlation between temperature and growth, development, and behavior.

Riparian Vegetation - Decreased light and temperature in steams can be a result of riparian vegetation (Cole, 1994). When the flow of water is slow, direct sunlight can significantly warm up the water, especially in the summer. In Pennsylvania, the average daily stream temperatures increased by 12 ^{o}C when flowing through an open area in direct sunlight but then decreased significantly during flow through 500 m of forest (Lynch et al., 1980).

However, during the winter, a lack of cover has the opposite effect and causes a decrease in temperature. Sweeney (1992) found that temperature changes of 2 to 6 ºC usually altered key life-history characteristics of some species. It has been observed that riparian forest buffers help to prevent changes in natural temperature patterns and also to mitigate the increases in temperature following deforestation.

Oxygen – Oxygen enters the water by absorption directly from the atmosphere and by plant photosynthesis (Mackenthun, 1969). Mountain streams that do not receive a lot of waste discharges are generally saturated with oxygen due to their shallow depth, constant motion, and large surface area exposed to the air.

Aquatic organisms only survive because of the dissolved oxygen which, at appropriate concentrations, enables them to reproduce and develop and gives them vigor. When oxygen levels are low, organisms experience stress and become less competitive in sustaining the species (Mackenthun, 1969).

Dissolved oxygen concentrations of 3 mg/L or less have been shown to interfere with fish populations for a number of reasons (Mackenthun, 1969). When the oxygen needed for chemical and biological processes exceeds the oxygen provided by re-aeration and photosynthesis, the fish will die. Dissolved oxygen concentrations will decrease and may even be depleted by slow currents, high temperatures, extensive growth of rooted aquatic plants, algal blooms or high concentrations of aquatic matter (Needham, 1969).

Pollution that depletes the stream of oxygen has a marked effect on stream communities (Odum, 1971). Major factors determining the amount of oxygen found in water are temperature, pressure, salinity, abundance of aquatic plants, and the amount of natural aeration from contact with the atmosphere (Needham, 1969). A level of 5 mg/L or higher of dissolved oxygen in water is the level associated with normal activity of most fish (Walburg, 1971).

In streams filled with trout, the dissolved oxygen concentration has been shown, by analysis, to be between 4.5 and 9.5 mg/L (Needham, 1969).

pH-value - Aquatic biota survive best when the water has a pH of 7, i.e. nearly neutral hydrogen ion activity. If the pH changes, either becoming more acidic or more alkaline, the stress levels increase and eventually species diversity and abundance decrease. In streams under the stresses of various human activities, the pH often becomes more acidic and many species suffer, as shown in Figure 10. (revised based on FISRWG, 1997).

One of the main causes for changes in the pH of aquatic environments is the increase in the acidity of rainfall (Schreiber, 1995). Some soils have the ability to buffer pH changes; however, those which cannot neutralize acid inputs cause environmental concerns.

Substrate - Substrate influences stream biota. Within one reach of a stream, different species and different numbers of species can be seen among microinvertebrate aggregations found in snags, sand, bedrock and cobbles (Benke et al., 1984; Smock et al., 1985; Huryn and Wallace, 1987).

The hyporheic zone is the area of substrate which is under the substrate-water boundary and is the main area for most benthic invertebrate species to live and reproduce. It may be only one centimeter thick in some cases or one meter thick in other cases. The hyporheic zone may form a large subsurface environment, as shown in Figure 11.

Stream substrates are composed of various materials, including clay, sand, gravel, cobbles, boulders, organic matter and woody debris. Substrates form solid structures that modify surface and interstitial flow patterns, influence the accumulation of organic materials, and provide for production, decomposition, and other processes (Minshall, 1984).

Sand and silt are considered to be the least suitable substrates for supporting aquatic organisms and provide for the fewest species and individuals. Rubble substrates have the highest densities and the most organisms (Odum, 1971). If woody debris, from nearby trees in forests and riparian areas, fall into the stream, the quantity and diversity of aquatic habitats is increased (Bisson et al., 1987; Dolloff et al., 1994).

Nutrients and Eutrophication – Nitrogen, phosphorus, potassium, selenium, and silica, are needed for plant growth. However, nitrogen and phosphorus, if found in surplus, may cause an increase in the rate of growth of algae and aquatic flora in a stream. This process is called eutrophication.

Eutrophication has been an environmental and ecological problem in China since the 1980s when the economy began to rapidly grow. If the excess organic matter is decomposed, it can result in oxygen depletion of the water, it also can have terrible aesthetic consequences the worst of which is the death of fish.

Eutrophication in lakes and reservoirs is indirectly measured as standing crops of phytoplankton biomass, usually represented by planktonic chlorophyll-a concentration. However, phytoplankton biomass is not generally the main component of plant biomass in smaller streams because the growth of periphyton and macrophytes, which live on the stream bed, is promoted by high substrate to volume ratios and periods of energetic flow. When there are decreased flows and high temperatures, excessive algal mats develop and oxygen is depleted due to eutrophication.

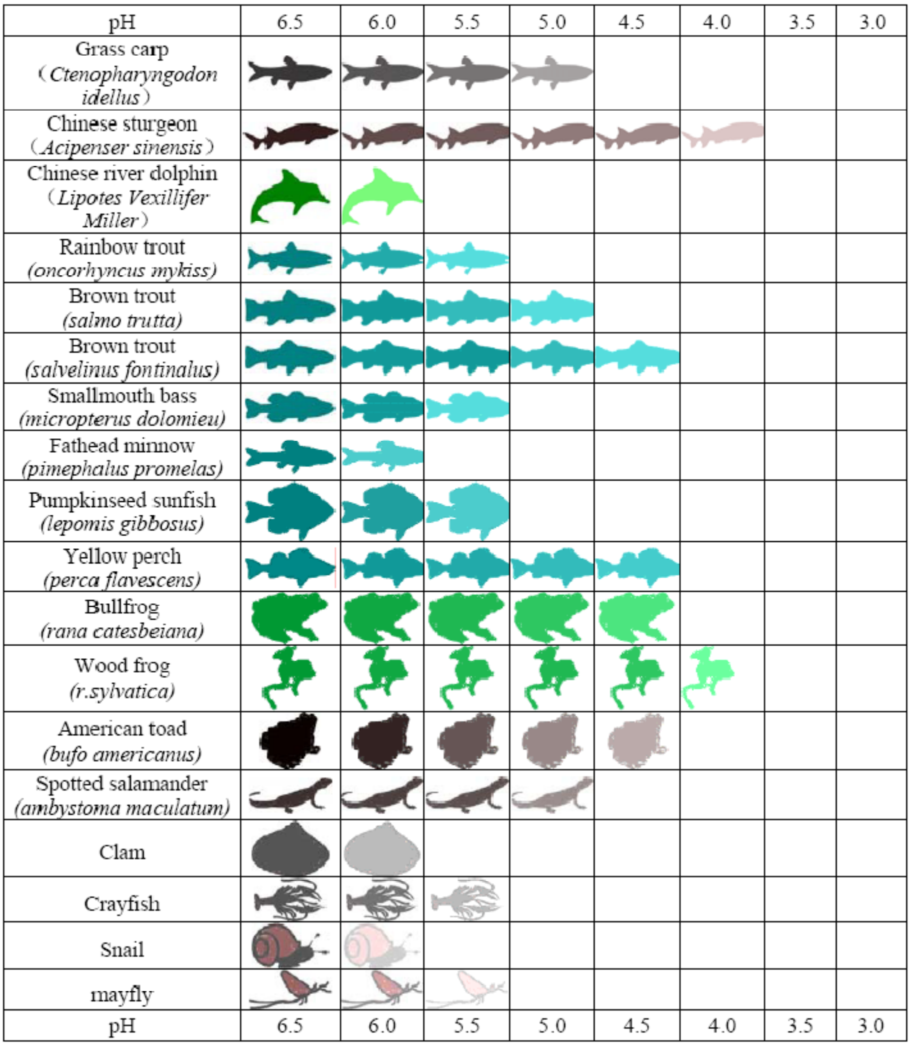

Figure 10. Effects of acid rain on some aquatic species. As acidity increases (pH decreases) in lakes and streams, some species are lost as indicated by the lighter colors (revised on the basis of FISRWG, 1997).

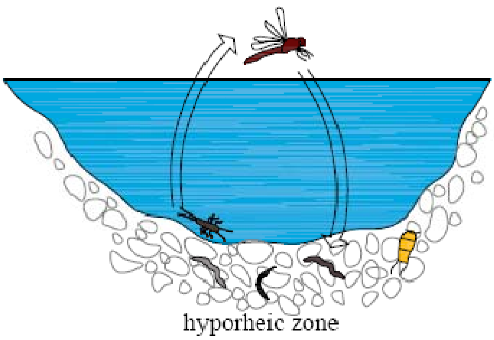

Figure 11. Schematic diagram of hyporheic zone.

1.4. Ecological Functions of Rivers

The main ecological functions of rivers are habitat, conduit, filter, barrier, source, and sink. Ecological restoration is done in order to enable river corridor functions to be effectively restored. However, the goals of restoration are not only to reestablish the structure or to restore a particular physical or biological process. Section 1. emphasizes matrix, patch, corridor, and mosaic as the most basic building blocks of physical structure at local to regional scales. Ecological functions, too, can be summarized as a set of basic, common themes that reappear in an ongoing range of situations.

Two characteristics are particularly important to the operation of stream corridor functions:

(1) Connectivity-This is a measure of the dimensions of a stream corridor and how far it continues (Forman and Godron, 1986). This attribute is affected by breaks in the corridor and between the stream and adjacent land uses. Transport of materials and energy and movement of flora and fauna are valuable functions promoted by a high degree of connectivity in a stream between its natural communities.
(2) Width- In stream corridors, this refers to the distance across the stream and its zone of adjacent vegetation cover. Width is affected by edges, community composition, environmental gradients and disturbances/ disruptions in adjacent ecosystems, including those with human activity. Average dimension and variance, number of narrows, and varying habitat requirements are some example measures of width (Dramstad et al., 1996).

1.4.1. Habitat Function

Habitat is a term used to describe an area where plants or animals (including people) normally live, grow, feed, reproduce, and otherwise exist for any portion of their life cycle. The important factors needed for survival such as space, food, water, and shelter are provided by the habitat. As long as the conditions are suitable, many species use river corridors to live, find food and water, reproduce, and establish viable populations. Population size, number of species, and genetic variation are a few measures of a stable biological community, which vary within known boundaries over time. Streams may positively affect these measures at different levels. Since corridors are linked to small habitat patches, they have a great value as habitats because they create large, more complex habitats with greater wildlife populations and higher biodiversity.

In general, stream corridors are habitats for plants, fish, invertebrates, and amphibians. For instance, the Fazi River is an urban stream in Taichong City, as shown in Figure 12. The river has gravel bed with alternative lentic and lotic waters, Although the river is seriously polluted in the upstream reaches several tens of species of macro-invertebrates, fish and birds are found in the river. Habitat functions differ at various scales, and an appreciation of the scales at which different habitat functions occur will help a restoration initiative succeed. The evaluation of a habitat at larger scales, for example, may make note of a biotic community's size, composition, connectivity, and shape. To help describe habitat over large areas at the landscape scale, the concepts of matrix, patches, mosaics, and corridors can be used. Migrating species can be provided with their favorite resting and feeding habitats during migration stopovers by stream corridors with naturally occurring vegetation.

Figure 12. The Fazi River, an urban stream in Taichong City, provides habitats for benthic invertebrates, fish and birds.

Some patches are too small for large mammals like the black bear which need great, unbroken areas to live in. However, these patches may be linked by wide stream corridors to create a large enough territory for bears. Assessing habitat function at small scales can also be viewed in terms of patches and corridors.

It is also at local scales that transitions among the various habitats within the river can become more important. Two basic types of habitat structure: interior and edge habitat can be found in stream corridors. Connectivity and width greatly influence the functions of habitats at the corridor scale.

A stream corridor provides a better habitat if it is wide and if it has greater connectivity. Changes in plant and animal communities can be caused by river valley morphology and environmental gradients, such as gradual changes in soil wetness, solar radiation, and precipitation. Species usually find ideal habitats in broad, unfractured, and diverse streams, rather than narrow and homogenous ones.

Factors such as climate, microclimate, elevation, topography, soils, hydrology, vegetation, and human uses, cause the habitat conditions within a river to vary. When planning to restore a stream, its width is of great importance to wildlife.

The size and shape of a stream corridor must be sufficiently wide for a species to populate. This must be considered when trying to maintain a certain wildlife species. If the corridor is too narrow, from the point of view of the species, it is as if there is a piece of the corridor missing. Riparian forests provide diversity not only in their edge and interior habitats but also offer vertical habitat diversity in their canopy, sub-canopy, shrub, and herb layers. Within the channel itself, riffles, pools, glides, rapids, and backwaters all provide different habitat conditions in both the water column and the streambed. These examples, all described in terms of physical structure, yet again show that there is a strong correlation between structure and habitat function.

1.4.2. Conduit Function

To act as a route for the flow of energy, materials, and organisms is known as the conduit function, as shown in Figure 13. A stream is foremost a conduit that was formed by and for collecting and transporting water and sediment. As well as water and sediment, aquatic fauna and other materials use the stream corridor as a conduit. Since there is movement across as well as along the stream and in many other directions, the corridor can be considered to have lateral and longitudinal conduit functions. If the stream corridor is covered by a closed canopy, then birds and mammals may cross over the stream through the vegetation. The food supply for fish and invertebrates may be enriched or increased by the movement of organic debris and nutrients from higher to lower floodplains. Corridors can act as pathways and habitats at the same time for migratory or highly mobile wildlife. The migration of songbirds from their wintering habitat in the neo-tropics to a summer habitat further north is made possible by rivers together with other, useful habitats.

After all, birds can only fly a certain distance before they need to eat and rest. For rivers to function effectively as conduits for these birds, they must be sufficiently connected and be broad enough to provide the habitat required for migratory birds. The migration of salmon upstream for spawning has been extensively investigated and is a well known example of the movement of aquatic organisms and interactions with the habitat.

A conduit to their upstream spawning grounds is very important to the salmon which mature in a saltwater environment. In the case of the Pacific salmon species the stream corridor depends on the nutrient input and biomass of dying fish and plentiful spawning in the upstream reaches. So, not only are conduits important for the movement of aquatic biota but also for the transport of nutrients from ocean waters upstream.

Stream corridors are also conduits for the movement of energy, which occurs in many forms. Heat is transported with flowing water along a stream, as shown in Figure 13. The potential energy of the stream is provided by gravity, which alters and carves the landscape. The corridor modifies heat and energy from sunlight as it remains cooler in spring and summer and warmer in the fall. Stream valleys move cool air from high to low altitudes in the evening, and, therefore, are effective air-sheds. The energy built up by the productivity of plants in a corridor is stored as living plant material and it moves into other systems by leaf fall and detritus.

Figure 13. A stream is a flow pathway for heat, water, and other materials, and organisms as shown for a small tributary of the Songhua River in northeast China.

Seeds may be carried for long distances by flowing water and then deposited. Whole plants may be uprooted, transported, and then deposited, still living, in a new area by strong floods.

Plants are also transported when animals eat and transport their seeds throughout different parts of the river. Some riparian habitats depend on a continuous supply and transport of sediment, although many fish and invertebrates can be harmed by excess fine sediment.

1.4.3. Filter and Barrier Functions

Stream corridors may act as filters, allowing selective penetration of energy, materials, and organisms, they may also act as a barrier to movement. In many ways, the entire stream corridor serves beneficially as a filter or barrier that reduces water pollution, minimizes sediment transport, and often provides a natural boundary to land uses, plant communities, and some less mobile wildlife species as shown in Figure 14.

Movement of materials, energy, and organisms perpendicular to the flow of the stream is most effectively filtered or barred, however, elements moving parallel to the stream corridor, along the edge, may also be selectively filtered. The movement of nutrients, sediment, and water over land is filtered by the riparian vegetation.

Dissolved substances, such as nitrogen, phosphorus and other nutrients, entering a vegetated river valley, are restricted from entering the channel by friction, root absorption, clay, and soil organic matter.

Edges at the boundaries of stream corridors begin the process of filtering. Initial filtering functions are concentrated into a tight area by sudden edges. These edges tend to be caused by disruptions and usually encourage movement along boundaries while opposing movement between ecosystems. On the other hand, gradual edges promote movement between ecosystems and increase filtering and spread it across a wider ecological gradient. Gradual edges are found in natural settings and are more diverse (FISRWG, 1997).

Figure 14. A stream functions as a boundary of land uses, plant communities, and some less mobile wildlife species.

1.4.4. Source and Sink Functions

Organisms, energy, and materials are supplied to the bordering area by rivers. Areas that function as sinks absorb organisms, energy, or materials from the surrounding landscape. A stream can act as both a source and a sink, as shown in Figure 15.

However, this is affected by the location of the stream and the time of year. Although they may sometimes function as a sink, when flooding deposits new sediment there, stream banks tend to act as a source, for example, of sediment to the stream.

Genetic material throughout the landscape is supplied by and moves through corridors, which at the landscape scale, act as conduits or connectors to many different patches of habitats.

Surface water, ground water, nutrients, energy, and sediment can be stored in stream corridors, which then act as a sink and allow materials to be temporarily stored in the corridor. Friction, root absorption, clay, and soil organic matter prevent the entry of dissolved substances such as nitrogen, phosphorus, and other nutrients into a vegetated stream corridor. Forman (1995) offers three sources and sink functions resulting from floodplain vegetation: 1) Decreased downstream flooding through floodwater moderation and/or uptake; 2) Containment of sediments and other materials during flood stage, and 3) Source of soil organic matter and water-borne organic matter.

Figure 15. A stream functions as a source providing organisms, energy, or materials to the surrounding landscape, and also as a sink absorbing organisms, energy, or materials from the surrounding landscape.

1.5. Dynamic Equilibrium

Even in the absence of human disruptions, the ecological characteristics of stream corridors, although consistent, are naturally and constantly changing in their structure, processes, and functions. Streams corridors show a dynamic form of stability. Stability is regarded as the capability of a system to prevail within a range of conditions. This phenomenon is referred to as *dynamic equilibrium*.

For a dynamic equilibrium to be preserved, the stream ecosystem must have an active series of self-correcting mechanisms. External disruptions and stresses may be moderated within a certain range of responses, by the ecosystem through these mechanisms, thus, preserving a self-sustaining condition. The threshold levels associated with these ranges are difficult to identify and quantify. If these levels are exceeded, the system can become unstable. A new steady state position may take a long time to become established, however this may be done if the stream is subjected to a series of adjustments.

Once the source of a disturbance is controlled or removed, a stream system can return to its working condition in a reasonable amount of time. The fact that ecosystems restore themselves after the removal of external stresses allows passive restoration. Removing the stress and allowing the ecosystem to recover by itself is an economical and effective method (see Chapter 9 for an example of ecosystem recovery after ammonia toxicity is reduced). However, following a profound disruption and alteration, the time needed for a stream corridor to heal itself could be several decades.

Even if a stream system does recover to equilibrium, it will be very different from before, and its ecological value may be greatly decreased. When an analysis by restoration practitioners shows that the recovery time will be long and questionable, they may decide that the use of active restoration techniques to reestablish a more operable channel form and biological community in a short time, may be viable. There are many difficulties with active restoration. Planning, designing, and implementing methods to regain the original state of dynamic equilibrium correctly are great challenges.

In some cases a disturbance may have such a great effect that the system may not be capable of recovery. In which case the stress must be removed and active restoration may be applied by repairing damage to the structure and function of the stream ecosystem. A stable ecosystem must have a combination of resistance, resilience, and capacity to recover. If it can keep its original form and functions, then it is resistant. The rate at which the ecosystem returns to its original form is known as its resilience. The resilience of an ecosystem, r_e, may be measured with the ratio of the stress, τ , over the time needed for the ecosystem to return its original form, T:

$$r_e = \frac{\tau}{T} \tag{1}$$

Recovery is the degree to which a system returns to its original condition after a disturbance. Natural systems are able to recover and restore stability following disruptions and disturbances. However, a system may not be able to recover if there is human activity as well as natural disturbances.

A system may change yet still remain stable and in a good condition. When a large stable system has small, local changes within it, it is described as having mosaic stability. A good example would be a riparian system greatly disturbed by a 100-year flood. If this were to occur in an area which was urbanized, then it would become a dangerous gap in a habitat that is already rapidly decreasing and would separate and isolate cause populations of rare amphibian species. However, that same system, in a less urbanized area may not be harmful to amphibians but may just represent a mosaic of constantly shifting suitable and unsuitable habitats in a naturally functioning, unconfined system. A landscape with mosaic stability is not likely to need restoration, whilst one without would urgently need restoring.

2. ECOLOGICAL STRESSES

Ecological stresses are defined as the disturbances that bring changes to river ecosystems. The ecological stresses are natural events or human-induced activities that occur separately or simultaneously. The structure of a system and its capability of carrying out important ecological functions may be changed by stresses, regardless of whether they act individually or in combination. One or more characteristics of a stable system may be permanently changed by a causal chain of events produced by a stress present in a river. For instance, land use change may cause changes of hydrological and hydraulic features of the river, and these changes may cause chages in sediment transportation, habitat, and ecology (Wesche, 1985).

Disturbances are not all of equal frequency, duration, and intensity and they may occur anywhere within the stream corridor and associated ecosystems. A large number of disturbances of different frequency, duration, intensity, and location may be caused by one single disturbance. Once people understand the evolution of what disturbances are stressing the system, and how the system reacts to those stresses, people can decide which actions are needed to restore the function and structure of the stream corridor.

Disturbance occurs within variations of scale and time. Changes brought about by land use, for example, may occur within a single year at the stream or reach scale (crop rotation), a decade within the stream scale (urbanization), and even over decades within the landscape scale (long-term forest management). Despite the fact that wildlife populations, such as the monarch butterfly, remain stable over long periods of time, they may fluctuate greatly in short periods of time in a certain area. Similarly, while weather fluctuates daily, geomorphic or climatic changes may occur over hundreds to thousands of years.

Although it is not observed by humans, tectonic motion changes the landscape over periods of millions of years. The slope of the land and the elevation of the earth surface are affected by tectonics, such as earthquakes and mountain creating forces like folding and faulting. Streams may alter their cross section or plan form in response to changes brought on by tectonics. Great changes in the patterns of vegetation, soils, and runoff in a landscape are caused by the quantity, timing, and distribution of precipitation. As runoff and sediment loads vary, the stream corridor may change.

2.1. Natural Stresses

Climatic change, desertification, floods, hurricanes, tornadoes, erosion and sedimentation, fire, lightning, volcanic eruptions, earthquakes, landslides, temperature extremes, and drought are among the many natural events that have a negative impact on the structure and functions of a river ecosystem. The relative stability, resistance, and resilience of an ecosystem determine their response to a disturbance.

Climate change may be illustrated with climate diagrams at meteorological stations. The climate diagram was suggested by Walter (1985). In the diagram, temperature is plotted on the left vertical axis and average total monthly precipitation on the right vertical axis. Temperature and precipitation are plotted on different scales. Walter (1985) used 20 mm/month as equivalent to 10^{o}C for the U.S. and Europe, but 100 mm/month is used as equivalent to 10^{o}C for a tropical rain forest. In this book, 30 mm/month equivalent to 10^{o}C for China. Very useful information, such as the seasonal fluctuation of temperature and precipitation, the duration and intensity of wet and dry seasons, and the percentage of the year in which the average monthly temperature is above and below 0^{o}C, is summarized in this climate diagram. When the precipitation line lies above the temperature line then, in theory, there should be enough moisture for plants to grow. The potential evapotranspiration rate will exceed the precipitation if the temperature line lies above the precipitation line. The more the temperature line moves up and away from the precipitation line, the drier the climate will be.

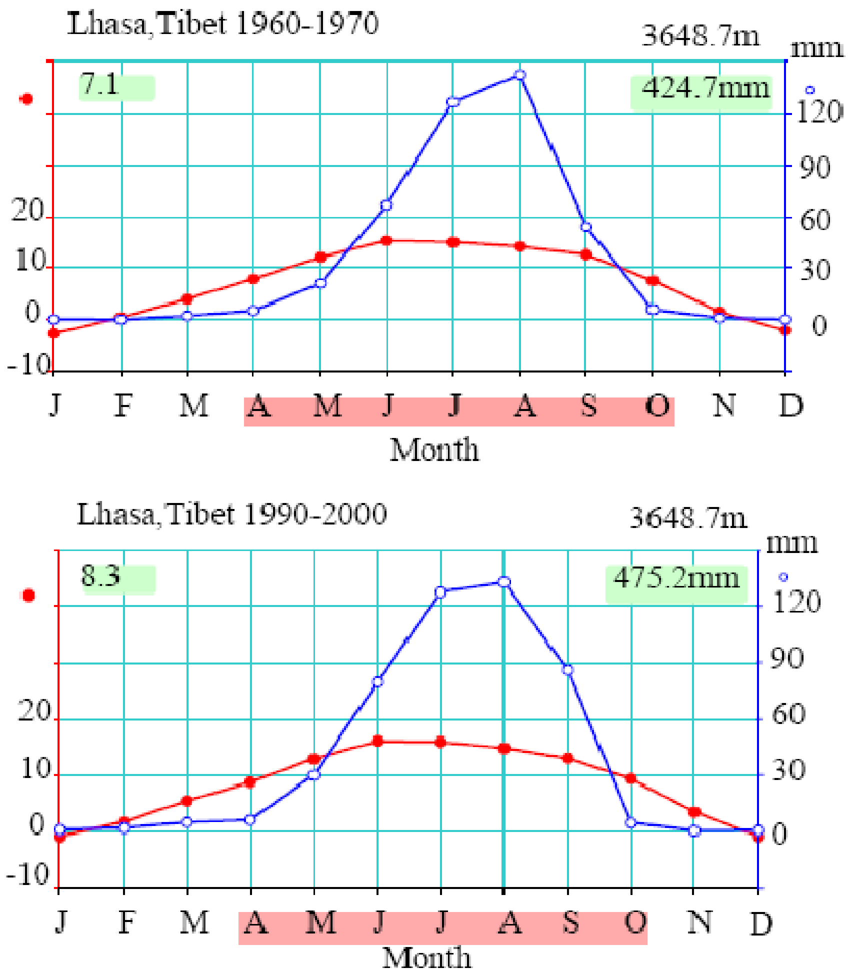

Figure 16. Climate diagrams for the Tibet Plateau (at Lhasa), in which average monthly temperature is plotted on the left vertical axis and average total monthly precipitation on the right vertical axis (10oC is equivalent to 30 mm of precipitation).

Figure 17. A serious a drought that started in 1997 and lasted for 6 years killed many poplar trees in the Kuye River basin in Inner Mongolia Autonomous Region, China. The trunks and branches die but the root system is still alive. The roots sprouted and new branches grew after the climate became wetter after 2004.

Global climate change can be represented by the climate change in the Tibet Plateau, as it is the third pole of the earth. The distributions of decade average monthly temperatures and total monthly precipitation are shown in each diagram in Figure 16. Between the 1960s and 1990s, the shape of the precipitation and temperature distributions has stayed the same. However, the average temperature has risen by about $1^{o}C$. The winter dry period which previously was from the beginning of October to the end of November, has extended from the beginning of October to the middle of December. The ecology of rivers around the world is affected by this climate change. As a consequence of global warming, continuous drought occurred in northern China. Figure 17. shows that many poplar trees in the Kuye River basin, which is a tributary of the Yellow River in northwest China, were killed by continuous drought from 1997 to 2003. The trunks and branches die but the root system is still alive. The roots sprouted and new branches grew after the climate became wetter after 2004.

Figure 18. shows that a sand dune in the Kubuqi desert in northwest China is moving toward a seasonal stream. The riparian forest has been damaged and some reptiles and amphibians have suffered. Desertification has become an environmental problem, which may be a consequence of deforestation and climate change.

Figure 18. A sand dune in Kubuqi desert in northwest China is moving toward a seasonal stream.

Figure 19. Wenjiagou Landslide in Mianzhu City buried the stream and vegetation.

Figure 20. (a) High sediment concentration in a stream in Taiwan, southeast China, which results in low transparency, low dissolved oxygen, and sediment coating the substrate; (b) Turbid seawater with high concentration of sediment impacts on fish and invertebrate communities.

A huge landslide may totally destroy the terrestrial and aquatic ecosystem of a river. Figure 19. shows the Wenjiagou Landslide in Mianzhu City, Sichuan, China, which was induced by the Wenchuan Earthquake on May 12, 2008. The total volume of the sliding body was 81 million m3. The stream and vegetation on slopes were buried underneath the 180m thick landslide. Both faunal and floral communities have been totally destroyed and the restoration needs a long period of time.

Erosion and sedimentation often are the direct cause of ecology impairment. Figure 20. (a) shows the high sediment concentration in a stream in Taiwan, southeast China, which causes a strong stress on the aquatic biocommunity. The sediment results from intensive soil erosion caused by a rainstorm. The high concentration results in low transparency, low dissolved oxygen, and sediment coating the substrate. Benthic animals and fish may be killed during the high concentration event. Figure 20. (b) shows the turbid seawater with a high concentration of sediment on the east coast of Taiwan. The sediment is transported into the ocean by debris flows and hyperconcentrated flows. Tidal current and waves bring the sediment onto the shore and bays, which impacts on fish and invertebrate communities.

Stream ecology is influenced by certain animal activities. For example, beavers build dams that cause ponds to form within a stream channel or in the floodplain. Figure 21. (a) shows that a couple of beavers skillfully use natures building materials and construct a wood dam with tree branches on the Spring Pond in Pennsylvania; and Figure 21. (b) shows the 3 m high beaver dam forms a pond, which provides a good habitat for fish and birds. Without any machines the beavers transported so much building materials and built the dam within several months. The landlord of the Spring Pond, Mr. R. Devries pronounced that there is no way for humans “could ever match their dam skills, their dam resourcefulness, their dam ingenuity, their dam persistence, their dam determination and their dam work ethic”.

Of course the dam construction by beavers disturbs the stream ecology. The pond kills much of the existing vegetation. Moreover, if appropriate woody plants in the floodplain are scarce, beavers extend their cutting activities into the uplands and can significantly alter the riparian and stream corridors. The sequence of beaver dams along a stream corridor may have major effects on hydrology, sedimentation, and mineral nutrients (Forman 1995). Silts and other fine sediments accumulate in the pond rather than being washed downstream. On the other hand the aquatic ecological conditions are improved by the beaver dams. Water from storm flow is held back, thereby affording some measure of flood control. Wetland areas usually form, and the water table rises upstream of the dam. The ponds combine slow flow, near-constant water levels, and low turbidity that support fish and other aquatic organisms. Birds may use beaver ponds extensively.

Although the Pennsylvania Department of Environmental Quality found that “dams of this nature are inherently hazardous and cannot be permitted”. The Department therefore orders “to restore the stream to a free-flow condition by removing all wood and brush forming the dams from the stream channel”. The beaver dam and the life of beavers on fish in their “reservoir” is a part of the ecology, which increases the diversity of habitats. The landlord Mr. R. Devries pronounced on behalf of the beavers that “the Spring Pond Beavers have a right to build their unauthorized dams as long as the sky is blue, the grass is green and water flows downstream. They have more dam rights than humans do to live and enjoy Spring Pond. If the Department of Natural Resources and Environmental Protection lives up to its name, it should protect the natural resources (Beavers) and the environment (Beavers' Dams).

Riparian vegetation, in general, tends to be resilient. Despite the fact that a flood may destroy a mature cottonwood forest, the conditions it leaves behind are usually those of a nursery, so a new forest can be established, and, thus, the riparian ecosystem is increased (Brady et al., 1985). Having developed characteristics such as high biomass and deep established root systems, the riparian forest systems have adapted to many types of natural stresses. Due to this adaptation, small and frequent droughts, floods, and other natural disruptions are of little consequence to the systems. When an unexpected serious stress occurs like fire, then the effect is only local and does not affect the community on a larger scale. However, the resilience of the system can be disrupted by widespread effects such as acid rain and indiscriminate logging and associated road building. Soil moisture, soil nutrients, and soil temperature can be critically changed by these and other disturbances, as well as other factors. Several tens of years are needed for the recovery of a system affected by widespread disturbance.

Figure 21. (a) A couple of beavers began to construct a dam with tree branches on the Spring Pond in Pennsylvania, U.S.; (b) The 3 m high beaver dam forms a pond, which provides a good habitat for fish and birds.

2.2. Human-Induced Stresses

Human-induced stresses undoubtedly have the greatest potential for introducing enduring changes to the ecological structure and functions of stream corridors. Physical disturbance effects occur at any scale from landscape and stream corridor to stream and reach, where they can cause impacts locally or at locations far removed from the site of origin. Activities such as flood control, road building and maintenance, agricultural tillage, and irrigation, as well as urban encroachment, can have dramatic effects on the geomorphology and hydrology of a watershed and the stream corridor morphology within it. The modification of stream hydraulics directly affects the system, causing an increase in the intensity of disturbances caused by floods. Chemically defined disturbance effects, for example, can be introduced through many activities including discharging sewage and wastewater (acid mine drainage and heavy metals) into the stream. Ecological disturbance effects are mainly to the result of the introduction of exotic species. The introduction of exotic species, whether intentional or not, can cause disruptions such as predation, hybridization, and the introduction of diseases.

For instance, bullfrogs have been introduced into the western U.S., They reproduce prodigiously and prey on numerous native amphibians, reptiles, fish, and small mammals and cause biological problems in the ecosystem. Altering the structure of plant communities can affect the infiltration and movement of water, thereby altering the timing and magnitude of runoff events.

Dams - Ranging from small temporary structures to huge multipurpose structures, human constructed barriers can have profound and varying impacts on stream corridors. The effect of disturbances resulting from barriers used for river impoundment on water quality, sediment transportation, and ecology are discussed extensively in Chapter 7. Barriers affect resident and migratory organisms in stream channels. Power plants may kill fish when they swim through the turbines. Figure 22. (a) shows that many birds are searching for dead fish at the outlets of a hydro-power plant in Korea, which were killed when they swam through the turbine; Figure 22. (b) shows that the Baozhusi Dam on the Bailong River in Sichuan Province has cut off the river flow. The stream ecology of the lower reaches has been greatly affected.

Figure 22. (a) Birds are searching for dead fish at the outlets of a hydro-power plant at which fish are killed when they swim through the turbine; (b) The Baozhusi dam on the Bailong River has cut off the flow and greatly affects the stream ecology in the lower reaches.

The dam blocks or slows the passage and migration of aquatic organisms, which in turn affects food chains associated with stream ecological functions. The Colorado River watershed is a 627,000-km^2 mosaic of mountains, deserts, and canyons. The watershed begins at over 4,000 m in the Rocky Mountains and ends at the Sea of Cortez. Many native species require very specific environments and ecosystem processes to survive. Under natural conditions, the basin's rivers and streams were characterized by a large stochastic variability in the annual and seasonal flow levels. This hydrologic variability was a key factor in the evolution of the basin's ecosystems. Today over 40 dams and diversion structures control the river system and result in extensive fragmentation of the watershed and riverine ecosystem.

Chanalization and water diversions - Like dams, channelization disturbs the stream ecology, by disrupting riffle and pool complexes needed at different times in the life cycle of certain aquatic organisms. The flood conveyance benefits of channelization and diversions are often offset by ecological losses resulting from increased stream velocities and reduced habitat diversity. Levees along rivers and diversion channels tend to replace riparian vegetation. The reduction in trees and other riparian vegetation along levees result in changes in shading, temperature, and nutrients. Hardened banks result in decreased habitat for organisms that live in stream sediments, banks, and riparian plants. Figure 23. (a) shows that the sediment banks of the lower Weihe River (the largest tributary of the Yellow River) have been replaced by stone walls in order to control floods and stabilize the channel. Figure 23. (b) shows that an urban stream in Beijing is reconstructed with concrete banks and channel bed for controlling seepage. Most of macro-invertebrates, reptiles, and amphibians have disappeared because the habitats for them have been covered with concrete.

Figure 23. (a) Hardened banks of the Weihe River; (b) Concrete banks and bed of an urban stream in Beijing.

Figure 24. Water flow in the Minjiang River is cut off due to water diversion.

Water diversion from rivers impacts the stream ecology, depending on the timing and amount of water diverted, as well as the location, design, and operation of the diversion structure. Figure 24. shows that the water flow in the Minjiang River, which is a tributary of the Yangtze River, is cut off due to water diversion. To exploit the hydro-power at low cost many low dams have been constructed on the river and the river water is diverted through pipelines and tunnels to the hydropower plants, which are located at several tens of kilometers downstream. Because all water has been diverted the reaches between the dams and the power plants became dry and all aquatic life were killed or seriously impacted. There are many such hydro-power plants on the rivers in southwest China, and, more hydro-power plants of such a type are being planned or are in construction. The management of stream ecology in southwest China is facing a serious challenge.

Fragmentation of habitat - Some river training works result in the fragmentation and isolation of habitats. Figure 25. shows the Yangtze River and numerous riparian lakes with different sizes. Naturally these lakes connected with the Yangtze River and formed a huge habitat in the past. Humans cut the connection for flood defense and aquatic farming, thus, fragmenting the habitat. The fragmentation of habitat has resulted in deterioration of the ecology and extinction of some species.

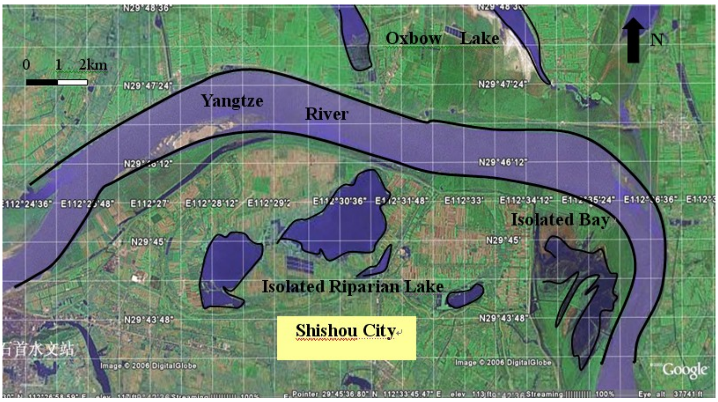

Figure 25. Isolation of riparian lakes along the Yangtze River results in fragmentation of habitat (Satellite image from the web http://earth.google.com)

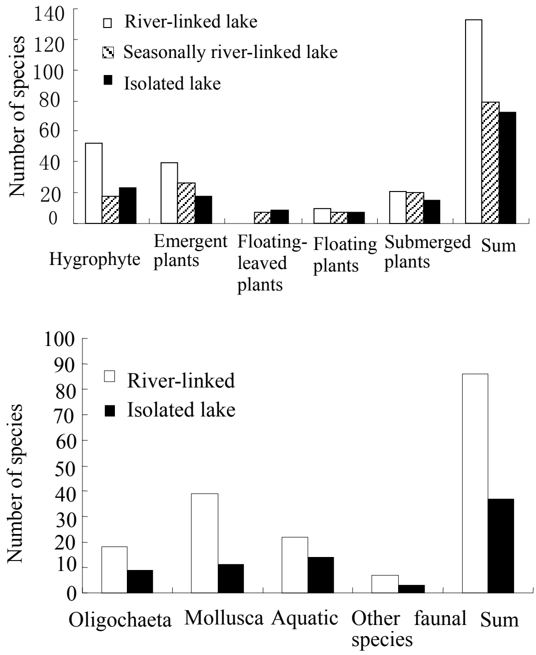

Figure 26. Comparison of species richness of aquatic plants and benthic invertebrates in isolated lakes and river-linked lakes in the middle and lower Yangtze River basin (after Wang and Wang, 2008).

Cut-off of riparian lakes from the Yangtze River stressed the complex ecosystem in the lakes and the river. Figure 26. shows a comparison of species richness of aquatic plants and benthic invertebrates in isolated lakes and river-linked lakes in the middle and lower Yangtze River basin (Wang and Wang, 2008). The connection of the isolated lakes with the Yangtze River was cut off in the past decades, which has resulted in continuous reduction of species. The cut-off also caused reduction of fish species. There are 101 fish species in the river-linked Poyang Lake but only 57 and 47 fish species in the isolated Honghu Lake and Zhangdu Lake.

An experiment was conducted in the Juma River in the suburbs of Beijing. In a backwater area of the river two experimental plots, a large one with area of 100 m^2 and a small one with area of 4 m^2, were separated from the neighboring water by steel sheet walls. The impact on benthic invertebrates was monitored for 6 months.

Figure 27. shows the variation of the total abundance and species richness in the two plots compared with those of the open water. In the first 40 days the differences between the three kinds of habitats are inconsistent.

Nevertheless, it becomes clear after 40 days that both abundance and species number in the closed plots reduced greatly. The abundance and richness in the small plot reduced to almost zero. In the large closed plot the abundance has reduced by two thirds and the species number reduced by one third (Duan and Wang, 2008). The results indicate that habitat fragmentation is a serious stress on the ecosystem and must be considered in ecological restoration of the river-lake complex system.

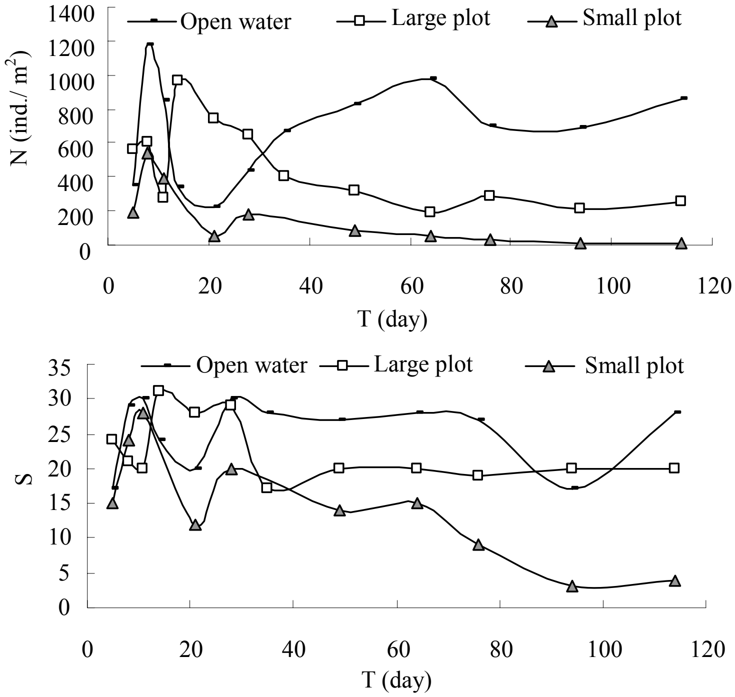

Figure 27. The number density, N, and species number, S, of benthic invertebrates in fragmented habitats (large plot area = 100 m2; small plot area = 4 m2) varying with the isolation time in comparison with that in the neighboring open water (after Duan and Wang, 2008).

Mining - Gold placer mining in rivers has become an extreme intensive disturbance to the stream ecology in southwest China. Figure 28.(a) shows placer mining in the Bailong River, which is a tributary of the Jialing River in Sichuan Province. People are removing bed gravel from the river for placer mining. The benthic invertebrate community is completely disturbed. Moreover, mercury is used in the process, which has also resulted in water pollution. Compared with gold mining, gravel mining is much more wide-spread. Since the 1980s, gravel mining has become a serious ecological stress in many rivers throughout China, as shown in Figure 28.(b). Gravel and coarse sand are mined for building materials. Gravel mining causes loss of habitat for benthic biocommunities and loss of spawning ground for many fish species. Lacking laws for controlling river sediment mining and attracted by great economic benefit, sediment mining has developed so quickly that almost all streams are stressed.

Surface mining also causes stresses on the river ecosystem. Exploration, extraction, processing, and transportation of coal, minerals, and other materials have had and continue to have a profound effect on stream corridors. Many rivers ecosystems remain in a degraded condition as a result of mining activities. Such mining activity frequently resulted in total destruction of the stream corridor. In some cases today, mining operations still disturb most or all of entire watersheds. Figure 29. shows a gold mine in the Henan Province in central China. Mercury was used to separate gold from the ore, therefore, mercury was also lost into streams. Present-day miners using suction dredges often find considerable quantities of mercury still resident in streambeds. Current heap-leaching methods use cyanide to extract gold from low-quality ores. This poses a special risk if operations are not carefully managed.

Figure 28. (a) Gold placer mining in the Bailong River, a tributary of the Jialing River in Sichuan; (b) Gravel mining for building materials from the Qingjiang River, a tributary of the Yangtze River.

Figure 29. A gold mine in the Henan in central China causes pollution of the Yihe River.

Pollution – Point source pollution from industry and diffuse pollution from agriculture (pesticides and nutrients) have the potential to disturb natural chemical cycles in streams, and, thus, to degrade water quality and impact the ecosystem.

Figure 30. (a) shows waste water from a factory discharged into the Jialing River, a large tributary of the Yangtze River in the Sichuan Province, and the seriously polluted river water. Riparian vegetation and animals near the outlet have been killed. The frog shown Figure 30. (b) has been killed due to the pollution. Heavy sewage discharge also causes stresses to the bio-communities.

Figure 31. (a) show the pollution of a stream flowing through the Dalian City, which is caused by uncontrolled sewage discharge. Fish and many other animals in the stream have been killed by the pollution. Figure 31. (b) shows eutrophication of the Lijiang River at Guilin, a famous tourism attraction for its beautiful landscape and streams. Sewage discharge from the city causes pollution of the river water and blooming of phytoplankton and macrophytes.

Toxic runoff or precipitates can kill streamside vegetation or can cause a shift to species more tolerant of polluted conditions. This affects habitat required by many species for cover, food, and reproduction. Aquatic habitat suffers from several factors.

Acid mine drainage can coat stream bottoms with iron precipitates, thereby affecting the habitat for bottom-dwelling and feeding organisms. Precipitates coating the stream bottom can eliminate places for egg survival. Fish that do hatch may face hostile stream conditions due to poor water quality.

Chemical disturbances from agriculture are usually widespread, nonpoint sources. Municipal and industrial waste contaminants are typically point sources and often chronic in duration.

Secondary effects, such as agricultural chemicals attached to sediments, frequently occur as a result of physical activities (irrigation or heavy application of herbicides). In these cases, it is better to control the physical activity at its source than to treat the symptoms within a stream corridor.

Figure 30. (a) Waste water from a factory is discharged into the Jialing River and seriously polluting the river water; (b) A frog is killed due to the pollution.

Figure 31. (a) Urban sewage polluting a stream flowing through the Dalian in northeast China; (b) Eutrophication of the Lijiang River at Guilin, a famous tourism attraction for its beautiful landscape and streams.

Urbanization – Urbanization in watersheds poses special challenges for stream ecological management. Recent research has shown that streams in urban watersheds have a character fundamentally different from that of streams in forested, rural, or even agricultural watersheds.

Impervious cover directly influences urban streams by dramatically increasing surface runoff during storm events by 2 to 16 times, with proportional reductions in ground water recharge (Schueler, 1995). Figure 32. conceptually shows the effects of different amounts of impervious cover on the water balance for a watershed.

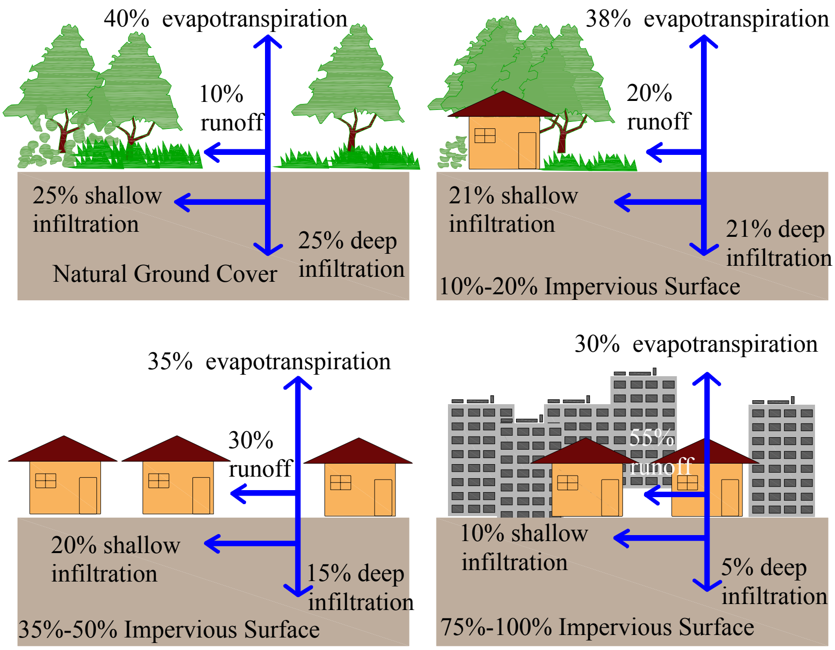

Figure 32. Effects of different amounts of impervious cover on the water balance for a watershed (after FISRWG, 1997) .

Figure 33. Riparian forest has been replaced by buildings and the banks are hardened with concrete in Guangyuan, Sichuan.

The unique character of urban streams often requires unique restoration strategies for the steam corridor. The peak discharge associated with the bankfull flow (1- to 2-year flood) increases sharply in magnitude in urban streams. Since impervious cover prevents rainfall from infiltrating into the soil, less flow is available to recharge ground water. Consequently, during extended periods without rainfall, baseflow levels are often reduced in urban streams (Simmons and Reynolds, 1982). Another modification unique to urban streams is the installation of sanitary sewers underneath or parallel to the stream channel. The water quality of urban streams during storm events is consistently poor. Urban storm water runoff contains moderate to high concentrations of sediment, carbon, nutrients, trace metals, hydrocarbons, chlorides, and bacteria (Schueler, 1987). Large woody debris is an important structural component of many small rivers, creating complex habitat structure and generally making the stream more retentive. In urban streams, the quantity of large woody debris found in stream channels is reduced due to the loss of riparian forest cover, storm washout, and channel maintenance practices (May et al., 1997). Many river crossings can become partial or total barriers to upstream fish migration, particularly if the streambed erodes below the fixed elevation of a culvert or a pipeline. The important role that riparian forests play in stream ecology often is diminished in urban watersheds since tree cover is often partially or totally removed along the stream as a consequence of development (May et al., 1997). Figure 33. shows a tributary of the Jialing River (a tributary of the Yangtze River) flowing through the Guangyuan that has been greatly affected by urbanization. Riparian forest has been replaced by residential buildings and the banks are hardened with concrete. The flow discharge in low the flow season has been reduced due to water diversion.

Agriculture and land-use change – Land-use change is the most common human-induced stress on the ecosystem. Agricultural activities have generally resulted in encroachment on stream corridors. Producers often crop as much productive land as possible to enhance economic returns; therefore, native vegetation is sacrificed to increase arable areas. As the composition and distribution of vegetation are altered, the interactions between ecosystem structure and function become fragmented. Vegetation removal from stream banks, floodplains, and uplands often conflicts with the hydrologic and geomorphic functions of stream corridors. These disturbances can result in sheet erosion, rill erosion, and gully erosion, reduced infiltration, increased upland surface runoff and transport of contaminants, increased bank erosion, unstable stream channels, and impaired habitat. Tillage and soil compaction interfere with the soil's capacity to partition and regulate the flow of water in the landscape, increase surface runoff, and decrease the water-holding capacity of soils. Tillage also often aids in the development of a hard pan, a layer of increased soil density and decreased permeability that restricts the movement of water into the subsurface. Disturbance of soil associated with agriculture generates runoff polluted with sediment, a major nonpoint source pollutant in the world. Pesticides and nutrients (mainly nitrogen, phosphorous, and potassium) applied during the growing season can leach into ground water or flow in surface water to stream corridors, either dissolved or adsorbed to soil particles. Improper storage and application of animal waste from concentrated animal production facilities are potential sources of chemical and bacterial contaminants to stream corridors. Tree removal decreases the quantity of nutrients in the watershed since approximately one-half of the nutrients in trees are in the trunks. Nutrient levels can increase if large limbs fall into streams during harvesting and decompose. Conversely, when tree cover is removed, there is a short-term increase in nutrient release followed by long-term reduction in nutrient levels. Removal of trees can affect the quality, quantity, and timing of stream flows. If trees are removed, from a large portion of a watershed, flow quantity can increase accordingly, and water temperature can increase during summer and decrease in winter. Many of the potential effects of land use change are cumulative or synergistic. Restoration might not remove all disturbance factors; however, addressing one or two disturbance activities can dramatically reduce the impact of those remaining. Simple changes in management, such as the use of conservation buffer strips in cropland or managed livestock access to riparian areas, can substantially overcome undesired cumulative effects or synergistic interactions.

Domestic livestock - Stream corridors are particularly attractive to livestock for many reasons. They are generally highly productive and provide ample forage. Husbandry development in a watershed has applied a unique stress on the ecosystem. For instance, the riparian vegetation succession from herbaceous to shrubs has been delayed or even stopped by grazing of livestock along the Ake River on the Qinghai-Tibet Plateau, as shown in Figure 34.(a). On the other hand, the activities of livestock have become an important element of the river ecology. Excrement of cattle provides the main nutrient for the grassland. The positive and negative effects of grazing of domestic livestock must be considered in any restoration strategy. In many cases livestock swimming in a stream can result in extensive physical disturbance and bacteriological contamination, as shown in Figure 34.(b).

Recreation and tourism - The amount of impacts caused by the recreation and tourism industry depends on stream hydrology, soil type, vegetation cover, topography, and intensity of use. Various forms of foot and vehicular traffic associated with recreational activities can damage riparian vegetation and soil structure. All-terrain vehicles, for example, can cause increased erosion and habitat reduction. At locations, reduced infiltration due to soil compaction and subsequent surface runoff can result in increased sediment loading to the stream (Cole and Marion 1988). In areas where the stream can support recreational boating, the system is vulnerable to additional impacts. Propeller wash and water displacement can disrupt and resuspend bottom sediment, increase bank erosion, and disorient or injure sensitive aquatic species, as shown in Figure 35.

Figure 34. (a) Grazing pressure has been increased due to development of husbandry in the Tibet-Qinghai Plateau; (b) Livestock swimming in a stream can result in extensive physical disturbance and bacteriological contamination.

Figure 35. Recreational boating, cruise tours, propeller wash, and accidental spills can degrade stream habitat.

Forestry-In addition to the changes in water, sediment, and nutrients loads to streams because of logging practices (i.e. land-use change), forestry may have other impacts of river ecosystems. Forest roads are constructed to move loaded logs to higher quality roads and then to a manufacturing facility.

Mechanical means to move logs to a loading area (landing) produce "skid trails." Stream crossings are necessary along some skid trails and most forest road systems and are in especially sensitive areas. Removal of topsoil, soil compaction, and logging equipment and log skidding can result in long-term loss of productivity, decreased porosity, decreased soil infiltration, and increased runoff and erosion. Spills of petroleum products can contaminate soils. Trails, roads, and landings can intercept ground water flow and cause it to become surface runoff.

2.3. Introduction of Exotic Species

Biologically defined disturbance effects occur within species (competition, cannibalism, etc.) and among species (competition, predation, etc.). These are natural interactions that are important determinants of population size and community organization in many ecosystems. Biological disturbances due to improper grazing management or recreational activities are frequently encountered. The introduction of exotic flora and fauna species can introduce widespread, intense, and continuous stress on native biological communities.

There are numerous examples worldwide of introduced species bringing about the extinction of native organisms. The most dramatic have involved predators. An extreme example is the deliberate introduction of the fish-eating Nile Perch *(Lates nilotica)* to Lake Victoria, in East Africa, causing the extinction of dozens of species of small endemic cichlid fish. Introduced cats, rats, and snakes have had a similar effect on island bird fauna (Dugeon and Corlett, 2004).

Exotic animals are a common problem in many areas in the U.S. and China. Species such as *Cambarus Clarkaij* have been introduced in many waters in south China. Without the normal checks and balances found in their native habitat in North America and Japan, *Cambarus Clarkaij* reproduces prodigiously and causes disturbance to the ecosystem. Figure 36. shows *Cambarus Clarkaij*. The species burrow in river levees and have caused many breaches and flooding disasters. The rapid spreading of the species has caused rice harvest reduction because the animals eat the rice paddies' root. In some places *Cambarus Clarkaij* has also caused prevalence of a disease.

Similarly introduction of the zebra mussel and bullfrog have imposed an intense stress on native biological communities in the western U.S. Without the normal checks and balances found in their native habitat in the eastern U.S., bullfrogs reproduce prodigiously and prey on numerous native amphibians, reptiles, fish, and small mammals.

Figure 36. Cambarus Clarkaij has been introduced in many waters in south China, resulting in ecological problems.

Golden mussel (*Limnoperna fortunei*) is an invasive filter species of macro-invertebrate. Originally the species comes from south China, which has spread to various regions, including Japan, Australia, Argentina, Thailand, India, Brazil and Europe (Darrigran et al., 2003). The species colonizes habitats with water temperature between 8-35 ℃, flow velocity less than 2 m/s, water depth less than 10 m with or without sunlight, dissolved oxygen higher than 1.0 mg/L, and pH higher than 6.4 (Morton, 1982; Márcia, 2006; Darrigran and Damborenea, 2006). Golden mussels have unprimitive byssus threads, which allow them to attach onto solid walls, especially human constructed water transfer tunnels and pipelines. Dense attachment of golden mussels in drink water transfer tunnels and pipelines results in macro-fouling (Yamada et al., 1997), causes high resistance to water flow and damage to pipeline walls. This along with dead mussels decay harms the surrounding water quality (Darrigran, 2002; Guan and Zhang, 2005).

Golden mussels have a high reproduction rate. Due to the favorable conditions in water diversion tunnels, cooling pipelines, pumps, and gate slots, golden mussel tends to colonize these habitats, ultimately leading to damage to these facilities. Figure 37. shows colonization of golden mussel in the water transfer tunnels in Shenzhen, southern China and attachment golden mussel on the surface of a concrete fragment. The density of golden mussel individuals is as high as to 20,000/m^2. Golden mussel invasion causes a serious challenge to water transfer projects that seek to solve issues such as the uneven distribution of water resources and the problem of water shortages in northern China. The presence of golden mussels results in quick and uncontrolled spread of the species.

Studies have been conducted to learn the golden mussels' biological characteristics in order to find effective and sustainable strategies to control its invasion (Morton, 1977; Wang, 1997; Darrigran and Damborenea, 2006; Liu et al., 2006; Li et al., 2007).

Figure 37. Colonization of golden mussel on concrete walls in a water transfer tunnel and attachment of golden mussel on a concrete fragment with high density.

Many measures have been suggested to control golden mussels (Xu et al, 2009), such as filter screen at the entrance of the water transfer systems (Darrigran, 2002), sand filter by using proper size distribution of sand (Lou and Liu, 1958), adjustment of flow velocity to restrain the attachment (Xiang, 1985), smooth pipe walls with anticorrosive coat to stop attach (Luo, 2006), temperature control, electromagnetic treating, and dissolved oxygen reducing (Morton,1975; Mcennulty et al., 2001), hot and dry environment (Iwasaki, 1977; Darrigran, et al., 2004), sealing up the pipelines to block oxygen from entering (Liu, 2006; Morton, 1977), hot water spraying to kill golden mussels (Morton, 1975), medicaments to kill golden mussels rapidly (Yamada et al., 1997), culture species of fishes preying golden mussels (Esteban et al., 2007), injecting certain steroids to disturb the normal spawning (John and Peter, 2001), and artificial or mechanical clearance during overhaul periods. Nevertheless, no effective and safe measures have been found to control the mussel. At present artificial or mechanical clearance during overhaul periods is the only effective and safe measure although it is expensive and causes intermittence of water supply.

Introduction of plant species may also cause stress on animal species. It was reported in the China Daily on Aug. 14, 2003, that the natural habitat of giant pandas, rare animals living mainly in China's Qinling Mountain range, is being threatened by larch forests (Figure 38). The story is as follows: "In recent years farmers in Sha-anxi province introduced larch trees because of its low planting cost, high survival rate and quick maturation into useful timber. However, the blind introduction of larch trees into the panda protection zones has severely depleted the groves of bamboo, the food staple of pandas. Larch trees have large crowns which can cause plants growing beneath them to die from a shortage of sunshine and water. And larch seeds, spread by birds and the wind, grow quickly. This can seriously impact the growth of surrounding plants. The introduction of larch into Qinling Mountain areas started in the early 1980s. In Foping County, the major home of pandas, more than 1,333 hectares of larch trees have been planted and the damaged bamboo forest cannot be recovered. The situation is also worsening in other protection zones, such as Changqing and Yangxian. The corridor linking panda clans in Ningshan and Foping has now been destroyed by larches. The habitat was decreased and the animals will suffer from the lack of food. Experts warned that the pandas will lose their home if the larch is not put under control, and the local government is expected to take measures soon."

Introduction of exotic species has inevitably occurred worldwide and this is accelerating following economic and ecologic. Compared with faunal species, introduction of floral species is quicker and more intensive because humans pay less attention to the negative impacts of the introduction. The introduction of exotic species, whether intentional or not, can cause disruptions such as hybridization and the introduction of diseases. Nonnative species compete with native species for moisture, nutrients, sunlight, and space and can adversely influence establishment rates for new plantings, foods, and habitat. In some cases, exotic plant species can even detract from the recreational value of streams by creating a dense, impenetrable thicket along the streambank.

Many exotic species have been introduced as consequences of human activities. For instance, at least 708 floral species and about 40 faunal species have been successfully introduced into China in the past century, among them several tens of species have caused ecological problems. A lot of money has been spent to remove these species. The most harmful species are *Eupatorium adenophorum*, *Eichoimia crassips, Ambrosia artemisia* L., and *Spartina alterniflora*.

Figure 38. Natural habitat of giant pandas is being threatened by introduction of larch trees and rapid development of larch forests.

Spartina alterniflora was introduced from the U.S. in 1980 to control coastal erosion and accelerate land creation in estuaries. The species may grow in salt marsh, because they tolerate periodical tidal inundation and resist wave erosion. The species colonize silt coasts very quickly and stabilize the coast with its dense roots. Nevertheless, the species dominate silt coasts and estuaries, resulting in a great reduction in biodiversity. Many invertebrates and fish cannot live in the shallow waters with *Spartina alterniflor*. The species has over to spread the neighboring coastal areas. Coastal areas and estuaries dominated by reed (*Phragmites communis trirn*) have been colonized and occupied by *Spartina alterniflor*. The fishery harvest has been significantly reduced. Figure 39.(a) shows *Spartina alterniflor* in the Yangtze River estuary.

Alien invasive species *Eupatorium adenophorum* originates from Mexico and was introduced into south China from south Asia in the 1940s. The species has spread quickly in southwest China and eliminated local species. The species is toxic and many cattle and sheep have been killed. The area occupied by the species has increased to about 30 million ha (Li and Wang, 2003). To remove the species from grassland is very difficult. Millions of dollars have been lost due to husbandry loss and control of spreading of the species in Sichuan and Yunnan Provinces. Figure 39.(b) shows the *Eupatorium adenophorum* in Yunnan Province in southwest China.

Eichoimia crassip was introduced to control eutrophication in streams and lakes. The species adsorb pollutants and nutrients in the water and may enhance the purification capacity of the stream or lake. Nevertheless, the species spread too fast and fishery and water surface recreation have been affected. Humans have to remove them from waters, which has caused economic losses up to several tens of millions of dollars. Figure 39.(c) shows *Eichoimia crassip* spreading quickly in a polluted stream in Beijing.

Ambrosia artemisia L. and *Ambrosia tritida* L. entered China in the 1930s and spread quickly since the 1980s and 1990s. The species produce a lot of pollen. In Shenyang in northeast China the density of pollen in air in 1987 was 38 times of that in 1983 because of introduction of the species. About 1.5% of the local people suffer from pollinosis. Millions of dollars have been lost due to introduction of the species. Figure 39.(d) shows the *Ambrosia artemisia* L. in northeast China.

Figure 39. (a) Spartina alterniflor in the Yangtze River estuary; (b) Eupatorium adenophorum in Yunnan Province in southwest China; (c) Eichoimia crassip spreading quickly in polluted waters; (d) Ambrosia artemisia L. in northeast China.

Introduction of exotic species is not always bad for the ecosystem. Hong Kong has become a paradise of exotic species and most of these species have been naturalized in the island. Hong Kong and the island of Dominica, in the Caribbean, probably had no inland plant species in common 500 years ago. Today they share more than a hundred weeds of human-dominated open habitats. The terms ‘alien’ is used to refer to species that originated elsewhere but have become established in Hong Kong. Although people have introduced many alien species by accident, others have been brought to a location deliberately, as crops, ornamentals, livestock, or pets. Not all introduced species are aliens. In fact, reintroduction of species to parts of their former range is an important conservation tool. Hong Kong’s total vascular plant flora of approximately 2,100 species includes at least 150 naturalized aliens: that is, species introduced from other parts of the world which have run wild in Hong Kong

(Dudgeon and Corlett, 2004). For faunal species, most of these aliens were brought to Hong Kong by people, but some have spread on their own. Some of these have established wild populations when they escaped, were abandoned or released. In Hong Kong the majority of introduced species are confined to those areas where human influence is strongest and most persistent. Indeed, in most residential and industrial areas, as well as the few sites still used for intensive farming, alien species dominate the biota. In contrast, recognizable aliens are rare or absent in most upland streams and hillside communities. Thus, the majority of aliens are found in those places where the native flora and fauna has already suffered most as a result of human activities. At present, the impact of the numerous alien plant and animal species established in Hong Kong is, in most cases, hard to distinguish from the direct impact of human activities on the habitats they occupy (Dudgeon and Corlett, 2004). The introduction of alien species into Hong Kong has increased the biodiversity and has resulted in no serious impacts on the local ecology. However, invasions by alien species are a potential conservation management problem that has received almost no attention in Hong Kong. Even if we ignore the risk posed by aliens to the ecology of Hong Kong, we have an obligation to ensure that the territory does not become a stepping-stone for invasion elsewhere.

3. Assessment of River Ecosystems

3.1. Indicator Species

Complete measurement of the state of a river ecosystem, or even a complete census of all of the species present, is not feasible. Thus, good indicators of the system conditions are efficient in the sense that they summarize the health of the overall system. The current value of an indicator for an impaired river ecosystem can be compared to a previously measured, pre-impact value, a desired future value, an observed value at an “unimpaired” reference site, or a normative value for that class of river ecosystems. For example, an index of species composition based on the presence or absence of a set of sensitive species might be generally correlated with water quality. If a river is polluted some species may be absent and the number of species may be less than that before the pollution. An index of indicator species itself provides no information on how water quality should be improved. However, the success of management actions in improving water quality could be tracked and evaluated through iterative measurement of the index. An indicator species group is defined as a set of organisms whose characteristics (e.g., number of species, presence or absence, population density, dispersion, reproductive success) are used as an index of attributes or environmental conditions of interest, which are too difficult, inconvenient, or expensive to measure for other species (Landres et al., 1988). The 1970s - 1980s is a peak interest period using aquatic and terrestrial indicator species for assessment of ecosystems. During that time, Habitat Evaluation Procedures (HEP) were developed by the U.S. Fish and Wildlife Service, and the use of management indicator species was mandated by law with passage of the National Forest Management Act in 1976. Since that time, numerous authors have expressed concern about the ability of indicator species to meet the expectations expressed in the above definition. Landres et al. (1988) critically evaluated the use of vertebrate species as ecological indicators and suggested that rigorous justification and evaluation are needed before the concept is used. Indicator species have been used to predict environmental contamination, population trends, and habitat quality. The assumptions implicit in using indicators are that if the habitat is suitable for the indicator it is also suitable for other species and that wildlife populations reflect habitat conditions. However, because each species has unique life requisites, the relationship between the indicator and its guild may not be completely reliable. It is also difficult to include all the factors that might limit a population when selecting a group of species that an indicator is expected to represent.

3.1.1. Selection of Indicator Species

Several factors are important to consider in the selection process of indicator species (FISRWG, 1997):

1) Sensitivity of the species to the environmental attribute being evaluated. When possible, data that suggest a cause-and-effect relation are preferred to mere correlation (to ensure the indicator reflects the variable of interest);
2) Indicator accurately and precisely responds to the measured effect. High variation statistically limits the ability to detect effects. Generalist species do not reflect change as well as more sensitive endemics. However, because specialists usually have lower populations, they might not be the best for cost-effective sampling. When the goal of monitoring is to evaluate on-site conditions, using indicators that occur only within the site makes sense. However, although permanent residents may better reflect local conditions, the goal of many riparian restoration efforts is to provide habitat for migratory birds. In this case, residents such as cardinals or woodpeckers might not serve as good indicators for migrating warblers;
3) Size of the species home range. If possible, the home range should be larger than that of other species in the evaluation area. Game species are often poor indicators simply because their populations are highly influenced by hunting mortality, which can mask environmental effects. Species with low populations or restrictions on sampling methods, such as threatened and endangered species, are also poor indicators because they are difficult to sample adequately;
4) Response uniformity in different geographic locations. Response of an indicator species to an environmental stress cannot be expected to be consistent across varying geographic locations or habitats. If possible, the response to a stress should be more uniform than that of other species in different geographic locations.

In summary, a good indicator species should be in the middle on the food chain to respond quickly and have relatively high stability, should have a narrow tolerance to stresses, and should be a native species (Erman, 1991). The selection of indicator species should be done through corroborative research.

3.1.2. Aquatic macro-invertebrates

Aquatic macro-invertebrates have been used as indicators of stream and riparian health for many years. Perhaps more than other taxa, they are closely tied to both aquatic and riparian habitat. Their life cycles usually include periods in and out of the water, with ties to riparian vegetation for feeding, pupation, emergence, mating, and egg laying (Erman, 1991).

It is often important to look at the entire assemblage of aquatic invertebrates as an indicator group. Impacts of stresses to a stream often decrease biodiversity but might increase the abundance of some species (Wallace and Gurtz, 1986). Using benthic macro-invertebrates is advantageous for the following reasons: 1) they are good indicators of localized conditions; 2) they integrate the effects of short-term environmental variables; 3) degraded conditions are easily detected; 4) sampling is relatively easy; 5) they are in the middle of the food chain and provide food for many fish of commercial or recreational importance; and 6) macro-invertebrates are generally abundant (Plafkin et al., 1989).

Field sampling of macro-invertebrates usually requires a combination of quantitative and qualitative collection methods. The sampling may be performed for one site in a 100 m stretch with representative areas of flow velocity, water depth, substrata composition, and hydrophyte growth. For a segment of an investigated stream, collections were made in areas with different current velocity, water depth, and different substrate sizes. At least three replicate samples were collected at each sampling site at appropriate depths of 0.15 m of the substrate with a kick-net (1 m ×1 m area, 420 μm mesh) if the water depth is less than 0.7 m. A D-frame dip net may be used to sample along stone surfaces and in plant clusters. If the water depth is greater than 0.7 m, samples may be collected with a Peterson grab sampler with an open area of 1/16 m^2. Replicate samples for each site are combined to form a composite sample, amounting to at least a minimum area of $1m^2$ (Duan et al., 2007). The cobbles sampled are generally scrubbed by hand to remove invertebrates and then discarded. The debris and invertebrates are rinsed vigorously through a fine sieve with a 300 μm mesh. Then the macro-invertebrates are taken from the debris and are placed in plastic sample containers and preserved in 10% formaldehyde in the field.

Environmental parameters, including substrate composition, water depth, water temperature, average current velocity, and dissolved oxygen concentration, are usually measured and recorded *in situ*. Growth and cover proportion of aquatic hydrophytes are also described. All macroinvertebrates are picked out of the samples and then identified and counted under a stereoscopic microscope in the laboratory. Macroinvertebrates are identified most to family or genus level except early-instar insects (Liu et al., 1979), and each species is assigned to a FFG based on the literature (Plafkin et al., 1989; Barbour et al., 1999).

3.1.3. Fish

Fish are also used as indicator species. Some management agencies use fish species as indicators to track changes in habitat condition or to assess the influence of habitat alteration on selected species. Habitat suitability indices and other habitat models are often used for this purpose, though the metric chosen to measure a species' response to its habitat can influence the outcome of the investigation. As van Horne (1983) pointed out, density or number of fish may be misleading indicators of habitat quality. Fish response guilds as indicators of restoration success in riparian ecosystems may be a valuable monitoring tool.

Hocutt (1981) states "perhaps the most compelling ecological factor is that structurally and functionally diverse fish communities both directly and indirectly provide evidence of water quality in that they incorporate all the local environmental perturbations into the stability of the communities themselves." The advantages of using fish as indicators are: 1) they are good indicators of longterm effects and broad habitat conditions; 2) fish communities represent a variety of trophic levels; 3) fish are at the top of the aquatic food chain and are consumed by humans; 4) fish are relatively easy to identify; and 5) water quality standards are often characterized in terms of fisheries. However, using fish as indicators is inconvenient because: 1) the cost of collection is high; 2) long term monitoring and a large number of samplings are needed to have reliable results and statistical validity may be hard to attain; and 3) the process of sampling may disturb the fish community. Electrofishing is the most commonly used field technique. Each collecting station should be representative of the study reach and similar to other reaches sampled; effort between reaches should be equal. All fish species, not just game species, should be collected for the fish community assessment. Karr et al. (1986) used 12 biological metrics to assess biotic integrity using taxonomic and trophic composition and condition and abundance of fish. The assessment method using fish as indicator has been studied and applied in many large rivers (Plafkin et al., 1989).

3.1.4. Birds and Mammals

Birds and mammals are used as indicator species for both terrestrial and aquatic ecosystems. Croonquist et al. (1991) evaluated the effects of anthropogenic disturbances on small mammals and birds along Pennsylvania waterways. They evaluated species in five different response guilds, including wetland dependency, trophic level, species status (endangered, recreational, native, exotic), habitat specificity, and seasonality. The habitat specificity and seasonality response guilds for birds were best able to distinguish those species sensitive to disturbance from those, which were not affected or benefited. Edge and exotic species were greater in abundance in the disturbed habitats and might serve as good indicators there. Seasonality analysis showed migrant breeders were more common in undisturbed areas, which, as suggested by Verner (1984), indicate the ability of guild analysis to distinguish local impacts.

In general the advantages of using birds and mammals as indicator species are: 1) they are good indicators of long-term effects and broad habitat conditions, including terrestrial and aquatic ecosystems; 2) they are at the top of the food chain; 3) they are relatively easy to identify; and 4) some restoration projects aim at restoration of endangered birds and mammals. The disadvantages are: 1) the cost of collection is high; 2) long term monitoring is needed to have reliable results; and 3) they are not sensitive to aquatic habitat conditions (e.g., hydrological changes or water pollution). Birds have been used as indicator species for ecological assessment of wetlands.

3.1.5. Algae

Algae communities are also useful for bioassessment. Algae generally have short life spans and rapid reproduction rates, making them useful for evaluating short-term impacts. Sampling impacts are minimal to resident biota, and collection requires little effort. Primary productivity of algae is affected by physical and chemical impairments. Algal communities are sensitive to some pollutants that might not visibly affect other aquatic communities. Algal communities can be examined for species, diversity indices, species richness, community respiration, and colonization rates. A variety of nontaxonomic evaluations, such as biomass and chlorophyll, may be used and are summarized in Weitzel (1979). Rodgers et al. (1979) describe functional measurements of algal communities, such as primary productivity and community respiration, to evaluate the effects of nutrient enrichment.

Although collecting algae in streams requires little effort, identifying for metrics, such as diversity indices and species richness, may require considerable effort. A great deal of effort may be expended to document diurnal and seasonal variations in productivity.

3.2. Metrics of Biodiversity

3.2.1. Richness and Abundance

If an indicator species group is selected, the ecosystem can be assessed by monitoring some variables of the indicator species group, including the species richness, S; the number density (or abundance), N, which is the total number of individuals per area; the biomass (the total weight of all individuals) per area; and the number of individuals per area for each species. Many parameters representing biodiversity of river ecosystems have been proposed. The species richness, S, is the most widely used indice (Magurran, 1988) and the most important characteristic of biodiversity:

$$S = \text{total number of species in the samples from a sampling site.} \qquad (2)$$

The ecological assessment and habitat conditions of streams may be mainly represented by the species richness. In general, the samples should be identified to species level for all species. Nevertheless, it is often not possible because to identity some species special instruments and experienced biologists are needed. In this case these species may be identified to genus level or family level. This does not affect the ecosystem assessment if the samples before and after the disturbance are examined by the same biologist and to the same level. A simple measure of richness is most often used in conservation biology studies because the many rare species that characterize most systems are generally of greater interest than the common species that dominate in diversity indices and because accurate population density estimates are often not available (Meffe et al., 1994). In general there are more species within large areas than within small areas. The relation between species richness, S, and habitat area, A, follows a power function formula (Ricklefs, 2001) :

$$S = cA^z \qquad (3)$$

where c and z are constants fitted to data. Analysis of species-area relations revealed that most values of z fall within the range 0.20-0.35 for birds and fish, and within the range 0.05-0.2 for benthic macro-invertebrates. For example, for the land-bird fauna of the West Indies, species richness increases from only 16 within an area of 10 km^2 to about 100 within an area of about 100,000 km^2. The relation between S and A is then (Ricklefs, 2001)

$$S = 10A^{0.24} \qquad (4)$$

The species richness increases with habitat area because habitat heterogeneity increases with the size of the area (and resulting topographic heterogeneity) of islands in the west Indies, and larger islands make better targets for potential immigrants from mainland sources of colonization. In addition, the larger populations on larger islands probably persist longer, being endowed with greater genetic diversity, broader distributions over area and habitat, and numbers large enough to prevent chance extinction. The fish community, like birds, also occurs in a large area of habitat and the sampling area must be large enough to a have reliable value of S. As a comparison, the macro-invertebrate community is more localized and needs much less sampling area for assessment of local ecosystems.

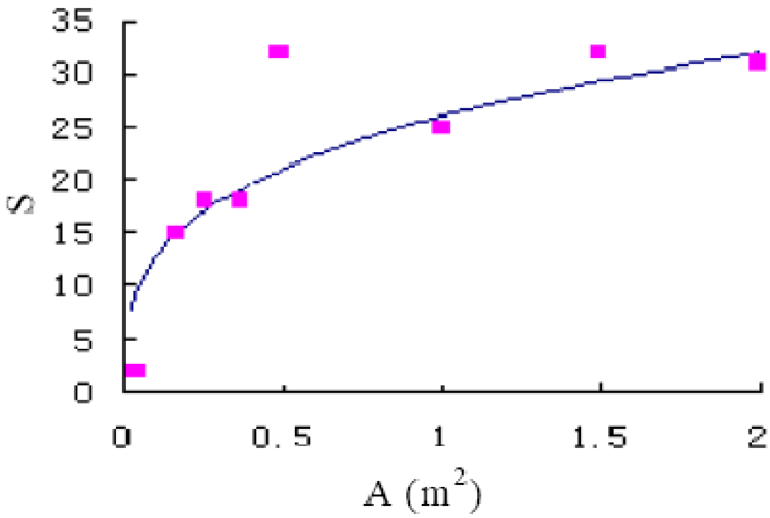

Figure 40. Relation of the species richness in a sample and the sampling area at each site.

If a river ecosystem with high heterogeneity of habitat is assessed with macro-invertebrates as indicator species, numerous sampling sites should be selected to represent different habitat conditions. For each sampling site the sampling area may be one or several m^2. The work load increases with the sampling area, therefore, ecologists prefer small sampling areas as long as a sufficient number of species can be sampled. Figure 40. shows the relation of the number of species in a sample and the sampling area at each site (Duan et al., 2007). The sampling area at each site should be at least 1 m^2 for a relatively reliable value of richness. The number density of individuals (abundance), *N*, is generally dynamic. If a bio-community colonizes a habitat at time t_0, the number density increases with time *t* and finally reaches equilibrium after a period of time. A differential equation describing the dynamic process of the number density growth is suggested (Ricklefs, 2001):

$$\frac{dN}{dt} = rN(1-\frac{N}{K}) \qquad (5)$$

in which r represents the intrinsic exponential growth rate of the population when its size is very small (that is, close to 0), and K is the carrying capacity of the environment, which represents the number of individuals that the environment can support. This equation is called the logistic equation. So long as N does not exceed the carrying capacity K, that is, N/K is less than 1, the number density continues to increase, albeit at a slowing rate. When N exceeds the value of K, the ratio N/K exceeds 1, dN/dt becomes negative, and the density decreases. K is the eventual equilibrium size of number density growing according to the logistic equation. Integration of the logistic equation yields

$$N = \frac{K}{1+\frac{K-N_0}{N_0}e^{-rt}} \qquad (6)$$

where N_0 is the number density of individuals at time t = 0. The logistic equation may be used for a species, e.g., black carp in Tongting Lake, or for a bio-community, e.g., benthic macro-invertebrates at a section of a stream. The abundance (density number) of a particular species reflects the balance between a large number of factors and processes, variations in each of which result in small increments or decrements in abundance. Population distribution models account for the evenness (equitability) of distribution of species, which fit various distributions to known models, such as the geometric series, log series, lognormal, or broken stick. In a large sample of individuals, species often distribute themselves normally over the logarithmic abundance categories.

3.2.2. Diversity Indices

Not all species should contribute equally to the estimate of total diversity, because their functional roles in the community vary, to some degree, in proportion to their overall abundance. Ecologists have formulated several diversity indices in which the contribution of each species is weighted by its relative abundance. Three such indices are widely used in ecology: Simpson's index, Margalef index and the Shannon-Weaver index. Simpson's index is

$$D=[\sum_{i=1}^{S}(\frac{n_i^2}{N^2})]^{-1} \tag{7}$$

in which n_i is the number of individuals of the i-th species, and N is the total number of individuals in the sample.

For any particular number of species in a sample (S), the value of D can vary from 1 to S, depending on the evenness of species abundances.

The Margalef index is defined as the total number of species present and the abundance or total number of individuals. The higher is the index, the greater the diversity. The Margalef index M is given by (Margalef, 1957):

$$M=(S-1)/\log_e^N \tag{8}$$

The Shannon-Weaver index, developed from information theory and integrating the species richness and evenness of the abundance distribution, is given by (Krebs, 1978):

$$H=-\sum_{i=1}^{S}\frac{n_i}{N}\ln\frac{n_i}{N} \tag{9}$$

The Shannon-Weaver Index provides no information on the total abundance of the bio-community. For instance, samples from two sites have the same number of species, the distributions are also the same but the density of individuals for site one is 10 ind/m^2 and for site two is 100 ind/m^2. Eq. (9) gives the same values of H. The difference in population density for the two cases is large, but it is not reflected by the values of H. Considering both the abundance and biodiversity, the following bio-community index is suggested (Wang et al., 2008b):

$$B = H \ln N = -\ln N \sum_{i=1}^{S} \frac{n_i}{N} \ln \frac{n_i}{N} \tag{10}$$

Macro-invertebrates census data from 9 sites along the East River in south China can be used to illustrate these different methods of presentation, as listed in Table 1. (Wang et al., 2008a). The East River is 562 km long and has a drainage area of 35,340 km^2. The river is one of the three major rivers of the Pearl River system – the largest system in South China. The Fenshuba Dam is a hydropower project on the river dividing the upper and middle reaches of the river and is 382 km from the river mouth. Figure 41. shows the variation of the species richness, *S*, number density of individual invertebrates, *N*, Shannon-Weaver index, *H,* and the bio-community index, *B,* from upper to lower reaches along the course. In general the richness, *S,* the density, *N*, Shannon-Weaver index, *H,* and the bio-community index, *B*, of benthic invertebrates reduce from the upper to the lower reaches. The Fenshuba Dam causes instantaneous fluctuation in flow discharge and velocity, which strongly impact the invertebrates. Therefore, only one species, *Palaemonidae*, which may survive the fluctuation, was found at the site downstream of the dam. The impact of velocity fluctuation becomes weak further downstream from the dam and exhibits no influence on the benthic invertebrates at a distance of 80 km from the dam.

Table 1. Species of benthic macro-invertebrates at the sampling sites along the East River

Sampling site	Species and the number of animals of each species per area (Figure within the parentheses is the number of individual animals of each species per square meter)
Shang-Pingshui	Baetidae (30); Melaniidae, *S.libertine* (23); Chironomidae (two species 16); *Ceratopsyche* sp.(7); *Aphropsyche* sp. (5); Elmidae (3); Corydalidae, *Protohermes* (3); Corbiculidae *C.nitens* (2); Polycentropodidae, *Neureclipsis* (2); Caenidae (1); *Helobdella* (1);
Feng-Shuba Dam	Palaemonidae (9)
Yidu	Leptophlebiidae, *Paraleptophlebia* (42); Chironomidae (21); Gomphidae (5); Siphlonuridae (4); Hydropsychidae (4); Leptophlebiidae, *Leptophlebia* (2); Decapoda (2); Hydrobiidae (2); *Semisulcospira* (1); Tipulidae, *Hexatoma* (1); Naucoridae (1); Corydalidae (1); Caenidae(1)
Wuxing	*Natantia* (44); *Bellamya* (10); *Branchiura* (3); *Radix* (2); *Melanoides* (2); Nepidae (1); *Limnodrilus* (1); Coenagrionidae, *Pseudagrion* (1); Leptophlebiidae, *Traverella* (1); Heptageniidae (1); Leptophlebiidae, *Paraleptophlebia* (1); Corbiculidae, *C.nitens* (1); Noteridae (1); Whitmania (1); Hirudinea sp$_1$.(1)；
Baipuhe	Palaemonidae (40); Palaemonidae, *Palaemon modestus* (12); Gomphidae (2); Macromiidae (2); *Semisulcospira* (2); *Branchiura* (2)
Huizhou	Chironomidae (3 species 11); Coenagrionidae (two species 6)；Branchiura (4); Paratelphusidae (1); *Ilydrolus* (1);Gomphidae (1); Platycnemididae (1); Ampullariidae (1)
Yuanzhou	0 (first sampling); Palaemonidae (9) (second sampling)
Dasheng	0 (first sampling); Palaemonidae (5) (second sampling)
Yequ Creek	Chironomidae (386); Simuliidae (18); Herpodellidae (4); Dytiscidae (3); *Branchiura* (3); Lumbriculidae (1); Psychodidae (1); Corduliidae, *Epitheca marginata* (1); Baetidae (1)

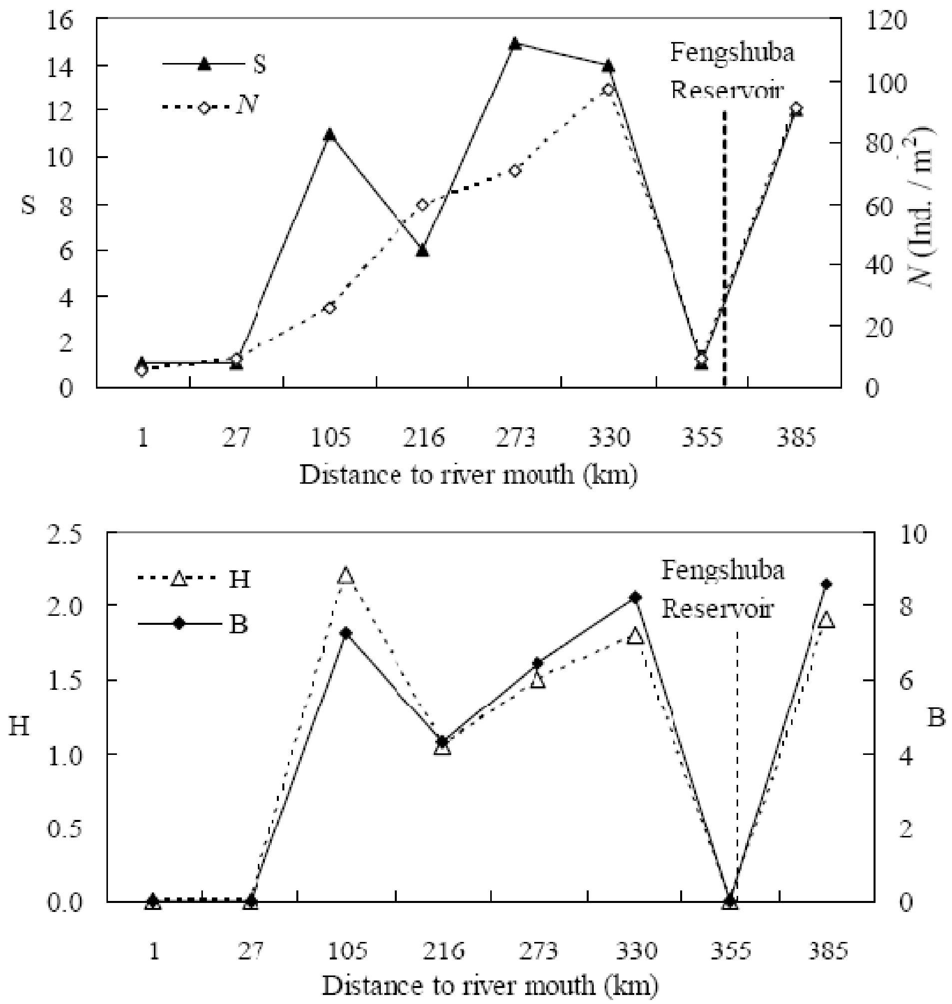

Figure 41. Species richness, S, number density of individual invertebrates per area, N, Shannon-weaver Index, H, and the bio-community index, B, as functions of distance to the river mouth.

In the lower reaches the channel has been regulated with relatively uniform width and the banks have been hardened with concrete and stones. Flow velocity in the channel is more uniform than the upper reaches and the substrate consists of only sand. The sand bed is compact, which provides no space for benthic animals to live and no shelter for the animals to escape current. The richness, number density, and biodiversity and bio-community indices in the lower reaches are very low or zero. Humans have reclaimed river bays, riparian lakes and wetlands, and sluggish and backwater zones, which caused loss of habitat and made formerly diversified habitats very uniform and unitary. In general, the biodiversity and bio-community indices are proportional to the diversity of habitats. The habitat loss and low diversity of habitats result in low bio-diversity and bio-community.

As indicated in the previous section biological diversity refers mainly to the number of species in an area or region and includes a measure of the variety of species in a community that takes into account the relative abundance of each species (Ricklefs, 1990). When measuring diversity, it is important to clearly define the biological objectives, stating exactly what attributes of the system are of concern and why (Schroeder and Keller, 1990). Different measures of diversity can be applied at various levels of complexity, to different taxonomic groups, and at distinct spatial scales.

Overall diversity within any given level of complexity may be of less concern than diversity of a particular subset of species or habitats. Measures of overall diversity include all of the elements of concern and do not provide information about the occurrence of specific elements. For example, measures of overall species diversity do not provide information about the presence of individual species, such as Chinese sturgeon, or species groups of management concern. Thus, for a specific ecological restoration project, measurement of diversity may be limited to a target group of special concern.

3.2.3. Alpha and Beta Diversities

Diversity can be measured within the bounds of a single community, across community boundaries, or in large areas encompassing many communities. Diversity within a relatively homogeneous community is known as alpha diversity, or local diversity. Usually the diversity indices obtained by examining the samples taken from one site are referred to as alpha diversity. Diversity between communities in a region, described as the amount of differentiation along habitat gradients, is termed beta diversity, or regional diversity. For instance, the total number of species from numerous sites along a stream is the regional diversity of the stream. Beta diversity may be large in river-lake connected habitats with high heterogeneity, because some species colonize stream habitat and very different species may live in the riparian lakes.

Noss and Harris (1986) note that management for alpha diversity may increase local species richness, while the regional landscape (gamma diversity) may become more homogeneous and less diverse overall. They recommend a goal of maintaining the regional species pool in an approximately natural relative abundance pattern. The specific size of the area of concern should be defined when diversity objectives are established.

A beta diversity index is given by the following formula:

$$\beta = \frac{M}{\frac{1}{S}\sum_{i=1}^{S} m_i} \tag{11}$$

in which M is the number of sampled habitats in a region, e.g., the middle reaches of the Yangtze River; m_i is the number of habitats, in which the *i-th* species is found; and S is the total number of species found at all sampling sites in the region. If the species in all sites are the same, or $m_i=M$, the beta diversity index is 1. If all species occur at only one site, $m_i=1$, the beta diversity index equals M. The total number of species, S, in the region is then the product of the average species richness by the beta diversity index. The ecological implication of beta diversity may be seen from the example of preliminary assessment of aquatic ecology of the source region of the Yellow River. The benthic macro-invertebrates were sampled at 8 sites with different environmental conditions in the source region of the Yellow River from Aug. 7 to Aug. 15, 2009. Figure 42. shows the location of 8 sampling sites. Samples were taken from 5 sites from the Yellow River and riparian waters. In addition, samples were taken from a small stream on the plateau, the Eling Lake and the Qinghai Lake. The sampling method is as follows: 1) In mountain streams with gravel beds, the gravels were washed and sieved with a kick-net with holes of 0.5 mm, and organic and inorganic detritus with macro-invertebrates collected. The detritus was subsequently placed on a white tray, and the invertebrates were collected. Invertebrate species were thereafter examined and identified to family or genus level under a microscope. The sampled area was 1.5 m^2 consists of three sub-sampling areas in order to reflect diversified ecological conditions. After sampling, macro-invertebrates and associated material were immediately preserved in ethanol and were subsequently processed and identified in the laboratory (Duan et al, 2009). There is little pollution and the water quality is very good. In general, the community of benthic invertebrates is different if the environmental conditions are different. The main environmental factors for benthic invertebrates are stream substrate, water depth, flow velocity, and water quality (Wang et al., 2009). At the site 1①the Zequ River is a tributary of the Yellow River with meandering channel. In its drainage area there are numerous swamps and rivulets with small but stable flow. The rivulets wriggle on vast meadows with grass coverage almost 100%. The site of streamlet represents the habitat type. Near the Yellow River by Kesheng town (site ②) there is an oxbow lake (site ③), which is abandoned channel of the Yellow River and has been cutoff from the river for a very long period of time. The site ④ is a riparian lake, which may connect with the Yellow River during high floods. The Dari bay ⑤ is a riparian wetland where the Yellow River flows from a normal channel to a very wide valley with shallow water. The main water flow has a deep channel, but plume of low sediment concentration drifts into the bay. The site ⑥ is a wetland by the Yellow River. The Eling Lake ⑦ is the source of the Yellow River with a capacity of 10 billion m^3. The pool level in the lake is not stable depending on the incoming water and operation of a hydropower station just below it. The water level had been rising since a month before the field investigation. The Qinghai Lake ⑧ has brackish water with low concentration of salt. It is near by the source of the Yellow River and represents a type of habitat in the region.

Table 2. lists the species of macro-invertebrates identified from the samples of each site with the number density (ind/m2) of each species in the parentheses (Wang et al., 2010). The taxa richness, or the number of species at each site, S, and the calculated biodiversity index B are listed in the table. Altogether 48 species of macro-invertebrates belonging to 24 families and 44 genera were identified. The average density and wet biomass of macro-invertebrates in the eight sampling stations were 360 ind/m2 and 2.3934 g/m2, respectively. Insects were predominant group, being 77.1% of the total in taxa number, 82.7% in density, and 88.6% in wet biomass. Figure 43. shows the representative species of macro-invertebrate in the sampling sites, which are dominant species or typical species at each site.

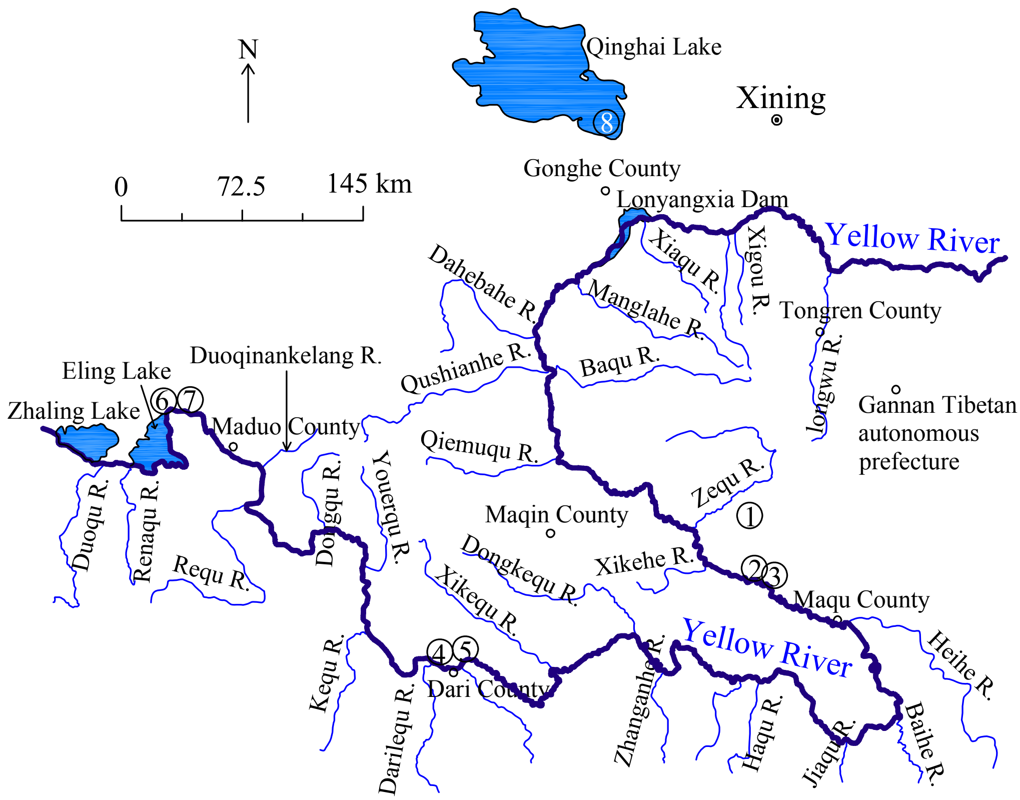

Figure 42. Location of the 8 sampling sites in the source region of the Yellow River.

The taxa richness *S* and the index *B* at each site are not high. In other words, the biodiversity of the sampling sites are not high. The streamlet and Qinghai Lake have only 6 species and both dominated by Amphipoda. The oxbow lake has the highest biodiversity, with 20 species and bio-community index about 12. In general, cobble, gravel and aquatic plants are the best substrates for benthic invertebrates. The oxbow lake, the isolated riparian lake at Dari, and the Yellow River channel at meanders have relatively stable environment and have multiple habitats with different substrates, therefore, they have high biodiversity. The Dari bay is an open shallow water connecting the Yellow River, its substrate consists of only fluid mud, and the sediment from the river drifting into the bay may change the fluid mud surface layer, thus it has relatively low biodiversity. Moreover, the species composition in the oxbow lake and riparian lake are quite different from the river and bay. These riparian waters are important in aquatic biodiversity.

The value of beta diversity was calculated for the source region of the Yellow River. The total number of sampled habitats is 8, so the value of *M* is 8 in Eq. (11). Calculation with the sampled species from the 8 habitats yields the beta diversity equal to 5.33, which is 66.7% of the maximum value.

As a comparison, field investigations were paid to the Juma River in the suburbs of Beijing from Shidu to Yesanpo with a length of about 70 km. The river is a mountain stream with beautiful landscapes and good aquatic ecology. The river reach from Shidu to Yesanpo is a main tourist attraction for Beijing people.

Table 2. Species composition of macro-invertebrates with densities (ind/m^2) in the parentheses

No.	Site	Species composition	*S*	*B*
1	Streamlet	*Limnodrilus grandisetosus* (2); Amphipoda (488); *Baetis* sp. (53); *Setodes* sp. (5); Tipulidae (1); *Eukiefferiella* sp. (9)	6	3.04
2	YR channel	Amphipoda (9); Acarina (2); *Baetis* sp. (3); *Cinygmula* sp. (4); *Ephemerella* sp. (30); *Leptonema* sp. (36); *Brachycentrus* sp. (3); Naucoridae (1); *Simulium* sp. (1); Culicidae (3); *Clinotanypus* sp. (1); *Eukiefferiella* sp. (9); *Orthocladius* sp. (2); *Cladotanytarsus* sp. (1); *Dicrotendipes* sp. (1); *Parachironomus* sp. (4); *Polypedilum* sp. (2)	17	9.86
3	Oxbow lake	Nematoda (300); *Aulodrilus pluriseta* (6); *Radix lagotis* (3); *Radix swinhoei* (12); *Hippeutis cantori* (9); *Hippeutis umbilicalis* (36); Amphipoda (93); Acarina (3); *Caenis* sp. (6); Dytiscidae (18); Elmidae (3); Corixidae (15); Pyralidae (336); *Procladius* sp. (15); *Chironomus* sp. (18); *Cryptochironomus* sp. (3); *Microchironomus* sp. (3); *Paratanytarsus* sp. (3); *Polypedilum braseniae* (3); *Xenochironomus* sp. (9)	20	11.84
4	Riparian lake	*Stylaria* sp. (1); *Limnodrilus* sp. (2); *Branchiura sowerbyi* (2); *Radix ovata* (4); Dytiscidae (8); Tipulidae (2); Culicidae (1); *Procladius* sp. (2); *Parametriocnemus* sp. (2); *Chironomus* sp. (10); *Cryptochironomus* sp. (1); *Endochironomus* sp. (1); *Paratanytarsus* sp. (17)	13	8.48
5	Dari bay	*Limnodrilus* sp. (3); Amphipoda (12); Tipulidae (1); *Psectrocladius* sp. (17); *Tvetenia* sp. (8); *Chironomus* sp. (10); *Polypedilum* sp. (13)	7	6.72
6	Eling Lake	no benthic animals	0	-
7	Riparian wetland	*Limnodrilus* sp. (1); Amphipoda (464); Culicidae (4); *Procladius* sp. (1); *Cricotopus* sp. (10); *Microchironomus* sp. (3); *Rheotanytarsus* sp. (75)	7	3.68
8	Qinghai Lake	Amphipoda (210); Culicidae (2); Ephydridae (2); *Cricotopus* sp. (105); *Eukiefferiella* sp. (19); *Chironomus* sp. (2)	6	5.32

Samples of benthic invertebrates were taken from 8 sites with different habitats, including gravel bed with turbulent flow, riparian wetland with lentic water, branch channel with low velocity flow, and pool behind weir. The substrates at the sampling sites were different, varying from gravel, cobbles, sand and macrophytes. The average taxa richness for the 8 different habitats was 19.4 and the highest taxa richness was 28. The total number of species was 54. The average value of index *B* for the 8 habitats was 10 and the highest value of *B* was 16. All the 8 habitats have high local biodiversity (alpha biodiversity). Nevertheless, the species compositions at different sites were rather similar. The beta diversity for the Juma River was only 2.7. The beta diversity for the source region of the Yellow River is two times of the Juma River. Ecological management or restoration in the region must base on an overall consideration of various habitats in the region.

Figure 43. Representative species of macro-invertebrate from the sites 1,2,3,4, 5 and 8.

3.2.4. Indices of Biotic Integrity

3.2.4.1. Karr's IBI

Fish represent the top of the aquatic food chain, and, thus, the quality and composition of the fish community comprises the best measure of the overall health of the aquatic community. This is because the fish community integrates the effects of the entire suite of physical, chemical, and biological stresses on the ecosystem. A fish community index should include at least one metric for each of the five attributes of fish assemblages (Simon and Lyons, 1995): species richness and condition, indicator species, trophic function, reproduction function, and individual abundance and condition.

Considering the foregoing considerations, Karr (1981) proposed and revised (Karr et al., 1986) the Index of Biotic Integrity (IBI) to evaluate stream quality at the fish community level. The Karr's IBI is comprised of 12 metrics to define fish community structure.

Table 3. Karr's Index of Biological Integrity (IBI) (after Karr et al., 1986)

Category	Metrics	Scoring Criteria		
		5	3	1
Species Richness and Composition	1. Total number of fish species	Expectations for metrics 1-5 vary with stream size and region		
	2. Number and identity of darter species			
	3. Number and identity of sunfish species			
	4. Number and identity of sucker species			
	5. Number and identity of intolerant species			
	6. Proportion of individuals as green sunfish	<5%	5-20%	>20%
Trophic Composition	7. Proportion of individuals as omnivores	<20%	20-45%	>45%
	8. Proportion of individuals as insectivorous Cyprinids	>45%	45-20%	<20%
	9. Proportion of individuals as piscivores (top carnivores)	>5%	5-1%	<1%
Fish Abundance and Condition	10. Number of individuals in sample	Expectations vary with stream size and region		
	11. Proportion of individuals as hybrids	0%	0-1%	>1%
	12. Proportion of individuals with disease, tumors, fin damage, skeletal anlmalies (DELT)	0-2%	2-5%	>5%

The index accounts for changes in fish community richness and allows for comparison of fish community composition with values for similar-sized streams. The applicability of the IBI concept has been demonstrated in a wide variety of streams types (Miller et al., 1988). As recommended by Karr et al. (1986), IBI metrics require adjustment for the region to which the index is applied. The basic components of Karr's index are listed in Table 3. It is recognized that stream size is an important factor when refining the IBI to a geographical region. The definitions of the twelve metrics are described as follows (Karr et al., 1986, Lyons, 1992):

Total number of species - The total number of species collected at a site, excluding hybrids and subspecies. The number of fish species supported by streams of a given size in a given region decreases with environmental degradation, if other features are similar.

Number of darter species – The total number of darter species (family *Percidae*) collected, excluding hybrids. Darters are small benthic species that tend to be intolerant of many types of environmental degradation. They are mainly insectivorous, and for many of them riffles or runs are preferred habitats. These species are sensitive to degradation, particularly as a result of their need to reproduce and feed in benthic habitats. Such habitats are degraded by channelization, siltation, and reduction in oxygen content.

Number of sunfish species – The total of sunfish species (family *Centrarchidae*), including rock bass (*Amobloplites rupertris*) and crappies (*Pomoxis* species), but excluding hybrids and black basses (*Micropterus salmoides*). Sunfish are medium sized, mid-water species, which tend to occur in pools or other shallow-moving water. Most, but not all, are tolerant of environmental degradation. All feed on a variety of invertebrates, although some larger adults may eat fish. Sunfish are included in the index because they are particularly responsive to the degradation of pool habitats and to other aspects of habitat such as instream cover.

Number of sucker species – The total number of sucker species (family *Catostomidae*) collected, excluding hybrids. Suckers are large benthic species that generally live in pools or runs, although a few species are common in riffles. Some species are intolerant of environmental degradation, whereas others are tolerant. Most species feed on insects, although a few also eat large quantities of detritus or plankton. Suckers are included in the index because many of these species are intolerant to degradation of habitat or chemical quality. Also, the longevity of suckers provides a multiyear integrative perspective.

Number of intolerant species – The total number of species, excluding hybrids, which are intolerant of environmental degradation, particularly poor water quality, siltation and increased turbidity, and reduced heterogeneity (e.g., channelization). Intolerant species are among the first to be decimated after perturbation to habitat or water quality, and the species identified in metrics 2-4 may be included in this group.

Proportion of individuals as green sunfish – In the Midwestern U.S., the green sunfish (*Lepomis cyanellus*) increases in relative abundance in degraded streams and may increase from an incidental to the dominant species. Thus, this metric evaluates the degree to which typically tolerant species dominate the community. In many other IBIs, tolerant species in the sample are listed and the proportion of tolerant individuals in the sample is computed and used as the metric in place of green sunfish.

Proportion of individuals as omnivores – The number of individuals that belong to species with an adult diet consisting of at least 25% (by volume) plant material or detritus and at least 25% live animal matter, expressed as a percentage of the total number of fish captured. By definition, omnivores can subsist on a broad range of food items, and they are relatively insensitive to the change in the food base of a stream caused by environmental degradation. Hybrids are included in this metric if both of the parental species are considered omnivores. The dominance of omnivores occurs as specific components of the food base become less reliable, and the opportunistic foraging habits of omnivores make them more successful than specialized foragers.

Proportion of individuals as insectivorous cyprinids – Cyprinids that belong to species with an adult diet normally dominated by aquatic or terrestrial insects, expressed as a percentage of the total number of fish captured. Although insectivorous cyprinids are a dominant trophic group in streams in the midwestern U.S., their relative abundance decreases with degradation, probably in response to variability in the insect supply, which in turn reflects alterations of water quality, energy sources, or instream habitat. In other regions the proportion of total insectivores to total individuals may provide better information for this metric with a resetting of the scoring criteria.

Proportion of individuals as piscivores (top carnivores) – The number of individuals that belong to species with an adult diet dominated by vertebrates (especially fish) or decapod crusteacea (e.g., crayfish, shrimp), expressed as a percentage of the total number of fish captured. Some species feed on invertebrates and fish as fry and juveniles. Hybrids are included in this metric only if both of the parental species are carnivores. Viable and healthy populations of top carnivores indicate a healthy, trophically diverse community.

Number of individuals in a sample – This metric evaluates populations and is expressed as catch per unit of sampling effort. Effort may be expressed per unit area sampled, per length of reach sampled, or per unit of time spent. In streams of a given size and with the same sampling method and efficiency of effort, poorer sites are generally expected to yield fewer individuals than sites of higher quality.

Proportion of individuals as hybrids – This metric is difficult to determine from historic data and is sometimes omitted for lack of data. Its primary purpose is to assess the extent to which degradation has altered reproductive isolation among species. Hybridization may be common among cyprinids after channelization, although difficulties in recognizing hybrids may preclude using this criterion with darters in addition to cyprinids. Sunfish also hybridize quite readily, and the frequency of their hybridization appears to increase with stream modifications.

Proportion of individuals with disease, tumors, fin damage, and skeletal anomalies (DELT) – The number of individual fish with skeletal or scale deformities, heavily frayed or eroded fins, open skin lesions, or tumors that are apparent from external examination, expressed as a percentage of the total number of fish captured. DELT fish are normally rare except at highly degraded sites.

Sampling of fish to determine these metrics is done on a reach basis. In Wisconsin, for example, a stream reach is defined as a minimum of 35 times the mean stream width based on at least 10 field measurements per site (Lyons, 1992). The results of the reach sampling are combined to define a sampling site.

3.2.4.2. IBI Examples

Karr's IBI concept has been adapted and modified from its midwestern U.S. beginnings for application throughout the world. Some IBIs simply adjust the scoring criteria as appropriate for their region of application, whereas other IBIs have combined new metrics with Karr's metrics. More than 40 fish metrics have been utilized in the various IBIs used in the U.S. (Limnotech, 2009).The IBI developed for Taiwan (Wu et al., 2005) is an example, where the majority of Karr's original metrics (with slight modifications) were applied, but the scoring criteria were modified (Table 4).

Other than the scoring criteria modifications, the main differences in the Taiwan IBI versus Karr's IBI are the use of all insectivores and consideration of numbers of hybrids or exotic species rather than the proportion of hybrids. Exotic species are species that are present in a region through introduction by man or have recent invasions that would not have been possible without human intervention. The total IBI scores then yield the following biological conditions categories: Non-impaired = 35-45, Slightly impaired = 23-34, Moderately impaired = 15-22, and Severely impaired = 0-14.

Table 4. Index of Biological Integrity (IBI) for Taiwan (after Hu et al., 2005)

Category	Metrics	Scoring Criteria		
		5	3	1
Species Richness and Composition	1. Total number of fish species	≥10	4-9	0-3
	2. Number of darter species	≥3	1-2	0
	3. Number of sunfish species	≥2	1	0
	4. Number of suckers species	≥2	1	0
	5. Number of intolerant species	≥3	1-2	0
Trophic Composition	6. Proportion of individuals as omnivores	<60%	60-80%	>80%
	7. Proportion of individuals as insectivores	>45%	45-20%	<20%
Fish Abundance and Condition	8. Number of individuals in sample	≥101	51-100	0-50
	9. Number of hybrids or exotic species	0	1	≥2

Karr's IBI and its many regional modifications for areas throughout the U.S. and around the world have generally been well calibrated to small "wadable" streams, but applications in larger rivers are less common (Lyons et al., 2001). Lyons et al. (2001) identified 7 IBIs developed for use in large rivers, and then developed IBIs for use in large rivers in Wisconsin. In this case large rivers are defined as having at least 3 km of contiguous river channel too deep to be effectively sampled by wading. Lyons et al. (2001) used fish assemblage data from 155 main-channel-border sites on 30 large warmwater rivers in Wisconsin (including 19 sites on the Mississippi River) to construct, test, and apply a large river IBIs. Fourteen sites were sampled more than once for a total of 187 samples. Watershed drainage areas for these sites ranged from 349 to 218,890 km^2. Lyons et al. (2001) used some of Karr's original metrics while adding several different metrics. A main difference is that instead of just considering the proportion of individuals (i.e. numbers-based metrics) the large river IBI also considers the proportion of fish by weight (i.e. biomass-based metrics). Such biomass-based metrics best reflect the amount of energy flow across trophic levels and functional groups, whereas numbers-based metrics indicate the diversity of pathways that energy could follow and the potential for intra- and inter-specific interactions (Lyons et al., 2001).

The large river IBI for southern Wisconsin is listed in Table 5. Definitions of some of the "new" metrics are given as follows (Lyons, 1992; Lyons et al., 2001):

Weight per unit effort – Weight (biomass) to the nearest 0.1 kg of fish collected per 1600 m of shoreline, excluding tolerant species.

Total number of native species - The total number of species collected at a site, excluding hybrids (which are common among sunfish and certain minnow species) and exotic species.

Total number of riverine species – Number of species that are obligate stream or river dwellers not normally found in lentic habitats.

Percent of individuals as simple lithophilous spawners - The number of individuals that belong to species that lay their eggs on clean gravel or cobble and do not build a nest or provide parental care, expressed as a percentage of the total number of fish captured. Simple lithophilous species need clean substrates for spawning and are particularly sensitive to sedimentation (embeddedness) of rocky substrates. Hybrids are included in this metric only if both of the parental species are simple lithophilous species.

The total IBI scores then yield the following biological conditions categories: Excellent = >80, Good = 60-79, Fair = 40-69, Poor = 20-39, Very Poor = <20. Lyons et al. (2001) found that the Wisconsin large river IBI was comparable to IBIs developed for use in large rivers in Ohio (including data for the Ohio River) and Indiana.

The fact that the IBI metrics in Table 5. reflect conditions on the Mississippi River and Ohio River indicate that these metrics might be a good beginning point for developing IBIs for the other large rivers of the world.

Table 5. Index of Biological Integrity (IBI) for large rivers in southern Wisconsin (after Lyons et al., 2001)

Metrics	Scoring Criteria		
	10	5	0
1. Weight of fish per unit effort	> 25 kg	10-25 kg	0-9.9 kg
2. Total number of native fish species	>15	12-15	0-11
3. Number of suckers species	>4	3-4	0-2
4. Number of intolerant species	>2	2	0-1
5. Number of riverine species	>6	5-6	0-4
6. Proportion of individuals with disease, tumors fin damage, skeletal anlmalies (DELT)	<0.5%	0.5-3%	>3%
7. Percent of individuals as riverine species	>20%	11-20%	0-10%
8. Percent of individuals as simple lithophilous spawners	>40%	26-40%	0-25%
9. Percent of insectivores by weight	>39%	21-39%	0-20%
10. Percent of round suckers by weight	>25%	11-25%	0-10%

3.2.4.3. Uses of the IBI

IBIs provide a valuable framework for assessing the status and evaluating the restoration of aquatic communities. IBIs encompass the structure, composition, and functional organization of the biological community. IBIs can be viewed as quantitative empirical models for rating the health of an aquatic ecosystem, providing a single, defensible, easily understood measure of the overall health of a river reach in question (Lyons et al., 2001). For example, IBIs can be used to quickly identify both high-quality reaches for protection and degraded sites for rehabilitation. While total IBI scores can provide the user with an indication that a stream fish community is potentially degraded by environmental stressors, the total score cannot provide the ability to identify which individual stressors are causing the response. The same total IBI score can be reached by an infinite combination of individual metric scores, each with its own environmental stressor. Thus, several researchers have focused not on the final IBI score, but rather on how the individual metrics can be used to describe the effects of anthropogenic stresses on the fish community (e.g., Manolakos et al., 2007, O'Reilly, 2007, Novotny et al., 2008, Bedoya et al., 2009). If relations between stresses and the fish community can be found ways to reduce these stresses and efficiently improve the fish community can be derived.

3.3. Bio-Assessment

3.3.1.. Rapid Bioassessment

Rapid bioassessment techniques are most appropriate when restoration goals are nonspecific and broad, such as improving the overall aquatic community or establishing a more balanced and diverse community in the river ecosystem (FISRWG, 1997). Bioassessment often refers to use of biotic indices or composite analyses, such as those used by the Ohio Environmental Protection Agency (Ohio EPA, 1990), and rapid bioassessment protocols (RBP), such as those documented by Plafkin et al. (1989).

The Ohio EPA evaluates biotic integrity by using an invertebrate community index that emphasizes structural attributes of invertebrate communities and compares the sample community with a reference or control community. The invertebrate community index is based on 10 metrics that describe different taxonomic and pollution tolerance relations within the macro-invertebrate community.

The rapid bio-assessment protocols established by the U.S. Environmental Protection Agency were developed to provide states with the technical information necessary for conducting cost-effective biological assessments (Plafkin et al., 1989). The RBP are divided into five sets of protocols, three for macroinvertebrates and two for fish, as shown in Table 6. The rapid bioassessment protocols RBP I to RBP III are for macroinvertebrates. RBP I is a "screening" or reconnaissance-level analysis used to discriminate obviously impaired and unimpaired sites from potentially affected areas requiring further investigation. RBP II and III use a set of metrics based on taxon tolerance and community structure similar to the invertebrate community index used by the State of Ohio.

Table 6. Five tiers of the rapid bioassessment protocols (after Plafkin et al., 1989.)

Level or Tier	Organism Group	Relative Level of Effort	Level of Taxonomy/ Where Performed	Level of Expertise Required
I	Benthic invertebrates	Low; 1-2 hr per site (no standardized sampling)	Order, family/field	One highly-trained biologist
Ⅱ	Benthic invertebrates	Intermediate; 1.5-2.5 hr per site (all taxonomy performed in the field)	Family/field	One highly-trained biologist and one technician
Ⅲ	Benthic invertebrates	Most rigorous; 3-5 hr per site (2-3 hr of total are for lab taxonomy)	Genus or species/ laboratory	One highly-trained biologist and one technician
V	Fish	Low; 1-3 hr per site (no fieldwork involved	Not applicable	One highly-trained biologist
Ⅵ	Fish	Most rigorous; 2-7 hr per site (1-2 hr are for data analysis)	Species /field	One highly-trained biologist and 1-2 technicians

Both are more labor-intensive than RBP I and incorporate field sampling. RBP II uses family-level taxonomy to determine the following set of metrics used in describing the biotic integrity of a stream: 1) Species richness, 2) Hilsenhoff biotic index (Hilsenhoff, 1982), 3) Ratio of scrapers to filtering collectors, 4) Ratio of Ephemeroptera/ Plecoptera/Trichoptera (EPT) and chironomid abundances, 5) Percent contribution of dominant taxa, 6) EPT index, 7) Community similarity index, and 8) Ratio of shredders to total number of individuals. RBP III further defines the level of biotic impairment and is essentially an intensified version of RBP II that uses species-level taxonomy. As with the invertebrate community index, the RBP metrics for a site are compared to metrics from a control or reference site.

3.3.2. Comparison Standard

With stream restoration activities, it is important to select a desired end condition for the proposed management action. A predetermined standard of comparison provides a benchmark against which to measure progress.

For example, if the chosen diversity measure is native species richness, the standard of comparison might be the maximum expected native species richness for a defined geographic area and time period. Historical conditions in the region should be considered when establishing a standard of comparison. If current conditions in a river are degraded, it may be best to establish the standard for a period in the past that represented more natural or desired conditions. In some cases historical diversity might have been less than current diversity due to changes in hydrology and encroachment of native and exotic riparian vegetation in the floodplain (Knopf, 1986). Thus, it is important to agree on what conditions are desired prior to establishing the standard of comparison.

For a hypothetical stream restoration initiative, the following biological diversity objective might be developed. Assume that a primary concern in an area is conserving native amphibian species and that 30 native species of amphibians have been known to occur historically in the watershed. The objective could be to manage the river ecosystem to provide and maintain suitable habitat for the 30 native amphibian species. River ecosystem restoration efforts must be directed toward those factors that can be managed to increase diversity to the desired level. Those factors might be the physical and structural features of the river ecosystem.

Diversity can be measured directly or predicted from other information. Direct measurement requires an actual inventory of the element of diversity, such as counting the amphibian species in the study area. Direct measures of diversity are most helpful when baseline information is available for comparing different sites. It is not possible, however, to directly measure certain attributes, such as species richness or the population level of various species, for various future conditions.

Predicting diversity with a model is generally more rapid than directly measuring diversity. In addition, predictive methods provide a means to analyze alternative future conditions before implementing specific restoration plans. The reliability and accuracy of diversity models should be established before their use.

3.3.3. Classification Systems

The common goal of classification systems is to organize variation. Classification systems include (FISRWG, 1997):

1. Geographic domain. The range of sites being classified varies from rivers of the world to local differences in the composition and characteristics of patches within one reach of a single river.
2. Variables considered. Some classifications are restricted to hydrology, geomorphology, and aquatic chemistry. Other community classifications are restricted to biotic variables of species composition and abundance of a limited number of taxa. Many classifications include both abiotic and biotic variables. Even purely abiotic classification systems are relevant to biological evaluations because of the important correlations (e.g., the whole concept of physical habitat) between abiotic structure and community composition.
3. Incorporation of temporal relations. Some classifications focus on describing correlations and similarities across sites at one, perhaps idealized, point in time. Other classifications identify explicit temporal transitions among classes, for example, succession of biotic communities or evolution of geomorphic landforms.
4. Focus on structural variation or functional behavior. Some classifications emphasize a parsimonious description of observed variation in the classification variables. Others use classification variables to identify types with different behaviors. For example, a vegetation classification can be based primarily on patterns of species co-occurrence, or it can be based on similarities in functional effect of vegetation on habitat value.
5. The extent to which management alternatives or human actions are explicitly considered as classification variables. To the extent that these variables are part of the classification itself, the classification system can directly predict the result of a management action. For example, a vegetation classification based on grazing intensity would predict a change from one class of vegetation to another class based on a change in grazing management.

Comparison of the degraded system to an actual unimpacted reference site, to the ideal type in a classification system, or to a range of similar systems can provide a framework for articulating the desired state of the degraded system. However, the desired state of the system is a management objective that ultimately comes from outside the classification of system variability.

3.3.4. Analyses of Species Requirements

Analyses of species requirements involve explicit statements of how variables interact to determine habitat or how well a system provides for the life requisites of fish and wildlife species. Complete specification of relations between all relevant variables and all species in a river system is not possible. Thus, analyses based on species requirements focus on one or more target species or groups of species.

In a simple case, this type of analysis may be based on an explicit statement of the physical factors that distinguish good habitat for a species (places where it is most likely to be found or where it best reproduces) from poor habitat (places where it is unlikely to be found or reproduces poorly).

In more complicated cases, such approaches incorporate variables beyond those of purely physical habitat, including other species that provide food or biotic structure, other species as competitors or predators, or spatial or temporal patterns of resource availability. Analyses based on species requirements differ from synthetic measures of system condition in that they explicitly incorporate relations between "causal" variables and desired biological attributes. Such analyses can be used directly to decide what restoration actions will achieve a desired result and to evaluate the likely consequences of a proposed restoration action. For example, an analysis using the habitat evaluation procedures might identify mast production (the accumulation of nuts from a productive fruiting season which serves as a food source for animals) as a factor limiting squirrel populations.

If squirrels are a species of concern, at least some parts of the stream restoration effort should be directed toward increasing mast production. In practice, this logical power is often compromised by incomplete knowledge of the species habitat requirements.

The complexity of these methods varies along a number of important dimensions, including prediction of habitat suitability versus population numbers, analysis for a single place and single time versus a temporal sequence of spatially complex requirements, and analysis for a single target species versus a set of target species involving tradeoffs. Each of these dimensions must be carefully considered in selecting an analysis procedure appropriate to the problem at hand.

3.4 Habitat Evaluation and Modeling

3.4.1. Habitat Diversity

Habitat evaluation is an important aspect of bio-assessment. Habitat has a definable carrying capacity, or suitability, to support or produce wildlife populations (Fretwell and Lucas, 1970). The capacity depends, to a great extent, on the habitat diversity. A habitat diversity index is needed to represent this characteristic. The physical conditions of stream habitat are mainly 1) the substrate; 2) water depth; and 3) flow velocity (Gorman and Karr, 1978). Different physical conditions support different bio-communities and diversified physical conditions may support diversified bio-communities. A habitat diversity index, H_D, is proposed as follows (Wang et al., 2008b):

$$H_D = N_h N_v \sum_i \alpha_i \tag{12}$$

where N_h and N_v are numbers for water depth diversity and velocity diversity, and α is the substrate diversity, which is different for different substrates. For water depth less than 0.1 m the habitat is colonized by species that like high concentrations of dissolved oxygen and plenty of light. For water depth larger than 0.5 m the habitat is colonized by species that like low light and dissolved oxygen. Many species may live in water with depths between 0.1-0.5 m. If a stream has three water areas: 1) shallow water, in which the water depth is in the range of 0~0.1 m; 2) mid depth water, in which the water depth is in the range of 0.1~0.5 m; and 3) deep water, in which water depth is larger than 0.5 m, and each of the three areas is larger than 10% of the stream water surface area, $N_h = 3$. If a stream has only shallow water and mid-depth water, and each of them is larger than 10% of the stream water surface area, $N_h = 2$. The value of N_h for other cases can be analogously obtained. For flow velocity less than 0.3 m/s the habitat is colonized by species that swim slowly. For velocity higher than 1 m/s the habitat is colonized by species that like high velocities. Many species live in the current between 0.3-1 m/s. If a stream has three water areas: 1) lentic area, in which the flow velocity is smaller than 0.3 m/s; 2) mid-velocity area, in which the flow velocity is in the range of 0.3~1 m/s; and 3) lotic area, in which the velocity is larger than 1 m/s, and each of the three areas is larger than 10% of the stream water surface area, $N_v = 3$. If a stream has only lentic and mid-velocity areas, and each of them is larger than 10% of stream water, $N_v = 2$. The value of N_v for other cases can be analogously obtained.

The selection of the critical values of water depth and velocity are determined by studying the habits of species, mainly of macro-invertebrates. It is found from field investigations that in the Yangtze River basin some species in the water depth between 0.1-0.5 m are different from those in shallower or deeper water. Similarly, some species living in the current range of 0.3-1 m/s are different from those in currents lower than 0.3 m/s or higher than 1 m/s. Beauger et al. (2006) reported that the highest species richness and density were found in various substrates where the velocity ranged between 0.3 and 1.2 m/s, and depths ranged from 0.16 to 0.5 m. Below 0.3 m/s the riverbed tends to be filled and not very productive, whereas above 1.2 m/s the current velocity acts as a constraint for most living material. Undoubtedly, at lower depths, vegetation and animals are disturbed by light, conversely at higher depths in which the primary productivity decreases, the bio-community is disturbed due to light attenuation. At lower and higher depths and velocities, only those species tolerant to the constraints may colonize the habitat.

Streambeds consisting of cobbles and boulders are very stable and provide the benthic macro-invertebrates diversified living spaces. Therefore, cobbles and boulders are associated with high habitat diversity. Stream flow over aquatic grasses has high velocity but the aquatic grasses generate a low velocity canopy, moreover, the aquatic grasses themselves are also habitat for some species. Thus, streams with aquatic grasses exhibit high habitat diversity. Some species may move and live within the fluid mud layer and consume the organic materials in the mud layer. The interstices in a fine gravel bed are small but sufficient for some species.

A sand bed is compact and the interstices between sand particles are too small for big benthic macro-invertebrates to move and live within them. If sand particles are moving as bed load the bed provides no stable habitat for animals. Therefore, moving sand is the worst habitat for benthic macroinvertebrates. Based on this discussion and field investigations of 16 streams, the α -values for various substrates are listed in Table 7. It is well known that large woody debris can substantially contribute to habitat quality in streams (Gippel, 1995; Abbe and Montgomery, 1996), and, thus, a more generally applicable listing of α -values should also include a value for stream substrates with large woody debris. However, large woody debris does not often occur in Chinese streams, therefore, a rating for large woody debris has not been determined and is not listed in Table 7. If a part of the streambed consists of one substrate and another part consists of the another substrate and both parts have areas larger than one tenth of the stream surface, the two α -values for the two kinds of substrates should be summed.

However, if sand or silt fills the interstices of gravel the α -value should be taken as for the substrate of sand or silt. If a streambed has three parts with different substrates: boulders and cobbles, aquatic grasses, and fluid clay mud, and each of the three parts is larger than one tenth of the total stream area, the sum of the α -values for the stream is $\sum_i \alpha_i = 6+5+3=14$. If the streambed is covered by moving sand and gravel or the bed is very unstable, $\sum_i \alpha_i = 0$.

Gorman and Karr (1978) also developed a Habitat Diversity Index combining the effects of substrate, velocity, and depth. They showed that fish species diversity and richness were strongly related to a combination of the effects of substrate, velocity, and depth. Their substrate classification is similar to that proposed here with the main differences being in the divisions of sediment sizes into the various classes, but a similar ordinal ranking is applied to the substrate material.

Table 7. Substrate diversity, α, values for different substrates (after Wang et al., 2008b)

Substrate	Boulders and cobbles (D＞200 mm)	Aquatic grass	Gravel (2-200 mm)	Fluid clay mud (D<0.02mm)	Silt (0.02 ~0.2 mm)	Sand (0.2~2 mm)	Unstable sand, gravel, and silt bed (0.02~20 mm)
α	6	5	4	3	2	1	0

They also developed class ranges for velocity and depth throughout a reach determined by a weighting of point measurements. The index applied here takes a simpler approach to considering the diversity of velocity and depth.

The biodiversity of streams depends not only on the physical conditions but also is affected by food availability and water quality. Food availability is very different for different species and should be studied separately.

Generally speaking, water pollution reduces the number of species but may not reduce the density of pollution-tolerant species. Water quality is not an inherent feature of a habitat and depends on human disturbances.

Therefore, water quality is not taken in the habitat diversity index. Water temperature also is an important factor for stream ecology. However, the temperature does not vary much in a reach of a stream unless a thermal discharge is present and it is not necessary to consider it in the analysis of local habitat diversity. When habitat across different zones with great temperature differences is studied, then temperature difference has to be considered in the analysis.

High diversity of habitat supports high diversity of bio-community, which may be illustrated with the sampling results of macro-invertebrates in several mountain streams in the Xiaojiang River basin in Yunnan Province in southwestern China. Figure 44. shows the relations between the habitat diversity, H_D, and the species richness, *S*, the Shannon Weaver index, *H*, and the bio-community index, *B*, for these streams.

In general, the higher is the habitat diversity, the higher are the species richness, the bio-diversity, and the bio-community index. However, the species richness, *S,* has the best relation with the habitat diversity clearly showing an increasing trend with habitat diversity. The bio-community index, *B,* also linearly increases with the habitat diversity. The Shannon-Weaver Index, *H*, increases with the habitat diversity, but the points around the $H_D \sim H$ curve is rather scattered.

The results suggest that the species richness, *S*, and bio-community index, *B*, are suitable ecological indicators for good habitat in streams that are not impaired by poor water quality. Similar results also were obtained from a study on the East River basin in Guangdong Province. Figure 45. shows the relations of the habitat diversity, H_D, with the Shannon-Weaver index, *H*, and bio-community index, *B,* for the East River.

The higher is the habitat diversity, the higher are the biodiversity and bio-community indices. The bio-community index, *B*, increases with habitat diversity, H_D, and the points of B-H_D relation are much closer to the curve than the relation of H-H_D.

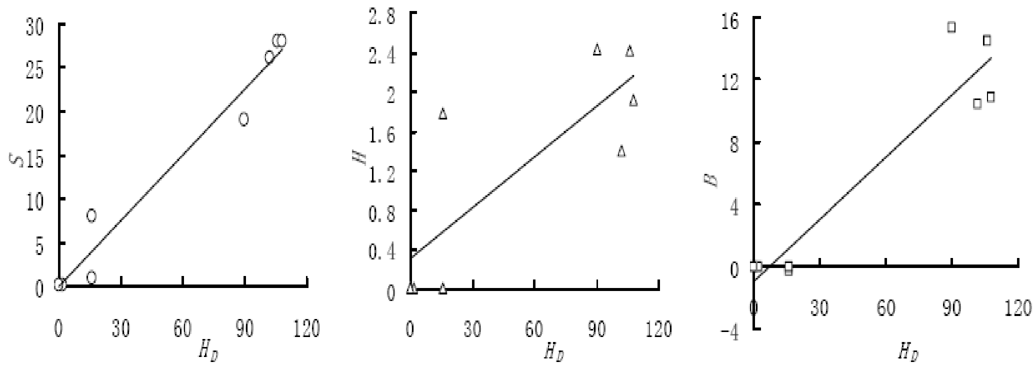

Figure 44. Species richness, S; Shannon-Weaver Index, H; and the bio-community index, B, as functions of the habitat diversity index, HD.

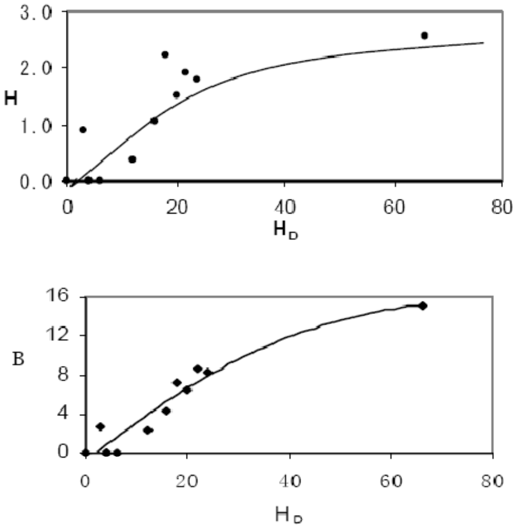

Figure 45. Relation between habitat diversity, HD, and Shannon-Weaver index, H (upper); and the relation between habitat diversity, HD, and bio-community index, B (lower).

3.4.2. Habitat Evaluation Procedure

The Habitat Evaluation Procedures (HEP) can be used for several different types of habitat studies, including impact assessment, mitigation, and habitat management. The HEP provides information for two general types of habitat comparisons—the relative value of different areas at the same point in time and the relative value of the same area at different points in time.

The HEP is based on two fundamental ecological principles—habitat has a definable carrying capacity to support wildlife populations, and the suitability of habitat for a given wildlife species can be estimated using measurements of vegetative, physical, and chemical characteristics of the habitat. The suitability of a habitat for a given species is described by a habitat suitability index (HSI) constrained between 0 (unsuitable habitat) and 1 (optimum habitat). HSI models have been developed and published (Schamberger et al., 1982; Terrell and Carpenter), the U.S. Fish and Wildlife Service (USFWS, 1981) also provides guidelines for use in developing HSI models for specific projects. HSI models can be developed for many of the previously described metrics, including species, guilds, and communities (Schroeder and Haire, 1993).

The fundamental unit of measure in The HEP is the Habitat Unit, computed as follows:

$$HU = AREA \times HSI \tag{13}$$

where HU is the number of habitat units (units of area), AREA is the areal extent of the habitat being described (in km^2), and HSI is the index of suitability of the habitat (dimensionless). Conceptually, an HU integrates the quantity and quality of habitat into a single measure, and one HU is equivalent to one unit of optimal habitat. The HEP provides an assessment of the net change in the number of HUs attributable to a proposed future action, such as a stream restoration initiative. A HEP application is essentially a two-step process—

calculating future HUs for a particular project alternative and calculating the net change as compared to a base condition.

3.4.3. Habitat Modeling

Many habitat evaluation models have been developed. The *Physical Habitat Simulation* **Model** was designed by the U.S. Fish and Wildlife Service primarily for instream flow analysis (Bovee, 1982). The model allows evaluation of available habitat within a study reach for various life stages of different fish species. The first component of the model is hydraulic simulation for predicting water surface elevations and velocities at unmeasured discharges (e.g., stage vs. discharge relations, Manning's equation, step-backwater computations). The second component of the model - habitat simulation integrates species and lifestage-specific habitat suitability curves for water depth, velocity, and substrate with the hydraulic data. Output is a plot of weighted usable area against discharge for the species and life stages of interest.

Riverine Community Habitat Assessment and Restoration Concept Model is based on the assumption that aquatic habitat in a restored stream reach will best mimic natural conditions if the frequency distribution of depth and velocity in the subject channel is similar to a reference reach with good aquatic habitat. Study site and reference site data can be measured or calculated using a computer model. The similarity of the proposed design and reference reach is expressed with three-dimensional graphs and statistics (Nestler et al., 1993, Abt, 1995). The model has been used as the primary tool for environmental analysis on studies of flow management for the Missouri River and the Alabama Basin.

SALMOD (Salmonid Population Model) is a conceptual and mathematical model for the salmonid population for Chinook salmon in concert with a 12-year flow evaluation study in the Trinity River of California using experts on the local river system and fish species in workshop settings (Williamson et al., 1993, Bartholow et al., 1993). The structure of the model is a middle ground between a highly aggregated classical population model that tracks cohorts/size groups for a generally large area without spatial resolution, and an individual-based model that tracks individuals at a great level of detail for a generally small area. The conceptual model states that fish growth, movement, and mortality are directly related to physical hydraulic habitat and water temperature, which in turn relate to the timing and amount of regulated stream flow.

Habitat capacity is characterized by the hydraulic and thermal properties, which are the model's spatial computational units. Model processes include spawning, growth (including maturation), movement (freshet-induced, habitat-induced, and seasonal), and mortality (base, movement-related, and temperature-related). The model is limited to freshwater habitat for the first 9 months of life; estuarine and ocean habitats are not included.

3.4.4. Suitability Indices

Suitability Indices are the core for habitat modeling, which may be illustrated for the Chinese sturgeon (Yi et al., 2007). The life cycle of the Chinese sturgeon in the Yangtze River mainly comprises spawning, hatching, and growth of 1 yr juvenile sturgeon. Brood fish seek suitable spawning sites; fertilized eggs adhere to stone and hatch after about 120 to 150 h. Whelp sturgeon drift with the current, and grow slowly in the lower reaches of the Yangtze River and river mouth. Juvenile sturgeons swim to the East China Sea and stay there until they reach maturity.

Therefore, analysis for the habitat quality of the Chinese sturgeon is based on basic requirements of spawning, hatching, and juvenile and adult sturgeon growth.

In habitat modeling variables which have been shown to affect growth, survival, abundance, or other measures of well-being of the Chinese sturgeon, are placed in the appropriate component. Ten aquatic eco-factors, which mainly influence the habitat of the Chinese sturgeon, are selected for the modeling as follows: 1) Water temperatures for adults and juveniles (V_1, °C); 2) Water depth for adults (V_2, m); 3) Substrate for adults (V_3); 4) Water temperature for spawning (V_4, °C); 5) Water depth for spawning (V_5, m); 6) Substrate for spawning and hatching (V_6); 7) Water temperature during hatching (V_7, °C); 8) Flow velocity during spawning (V_8, m/s); 9) Suspended sediment concentration during spawning (V_9, mg/l); and 10) The amount of eggs-predating fish in the studied year in comparison to a standard year (V_{10}). The suitable ranges and the Suitability Index (*SI*) curves of the ten main eco-factors are determined based on biological research. By analyzing these eco-factors, a habitat assessment model is developed which combines these factors and can be used for assessing habitat changes caused by human activities and hydraulic processes. The habitat suitability function for the Chinese sturgeon mainly considered the suitability for juvenile and adult fish growth, spawning, and hatching.

Habitat Suitability Index:

$$HSI = \min(C_{Ad},\ C_{Sp},\ C_{Ha}) \tag{14}$$

in which C_{Ad} represents the suitability for juvenile and adult growth, given by

$$C_{Ad} = \min(V_1, V_2, V_3) \tag{15}$$

C_{Sp} represents the suitability for spawning

$$C_{Sp} = \min(V_4,\ V_5,\ V_6) \tag{16}$$

C_{Ha} represents the suitability for hatching

$$C_{Ha} = V_{10} \bullet \min(V_6,\ V_7,\ V_8,\ V_9) \tag{17}$$

where V_1 – V_{10} are the ten factors. The *SI* curve quantifies physical habitat such as water temperature, flow velocity, and suspended sediment concentration. The habitat suitability ranges from unsuitable (0) to optimal habitat suitability (1). The intermediate values represent the suitability range based on a specified hydraulic variable.

Biological studies discovered that adult sturgeon distribution, spawning time, and spawning site selection by brood fish, are mainly influenced by water temperature (V_1, V_4), water depth (V_2, V_5) and substrate (V_3, V_6). The main eco-factors which influence hatching are water temperature (V_7), flow velocity (V_8), substrate (V_6), suspended sediment concentration (V_9), and the amount of the eggs-predating fish (V_{10}). Water temperature is an essential factor for hatching; flow velocity influences the distribution of eggs and their cohesiveness on the river bed. Excessive suspended sediment concentration may cause sturgeon eggs to debond, which then affects fertilization and hatching.

According to Chang (1999), 90% of sturgeon eggs suffer predation. The data sources used to develop the SIS are listed in Table 8, and the *SI* curves are shown in Figure 46. The value of V_{10} (the ratio of estimated brood sturgeon to eggs-predatory fish) is not shown in the figure, because it depends on the physical conditions and the number of the eggs-predatory fish in the previous year. In the modeling the value of V_{10} is assumed equal to 1.0, i.e. the amount of eggs-predatory fish is the lowest in the record.

Table 8. Eco-factors for Chinese Sturgeon (after Yi et al., 2007)

Variables	Eco-factors	Results of Previous Research
V_1	Water Temperature (Adults and Juveniles)	The Chinese sturgeon can survive temperatures between 0 – 37°C; 13 – 25°C is suitable for growth, and 20 – 22°C is optimum. The sturgeon becomes anorexic and stops growing when temperatures fall to 9 – 6°C (NERCITA, 2004a). Research results indicate that the Chinese sturgeon grows well under a wide range of temperatures; feeding has been recorded from 8 – 29.1°C (Guo and Lian, 2001). Yan (2003) found that Chinese sturgeon prefer tepid water, anorexia results and growth almost stops when temperatures are < 6°C and > 28°C, growth rate slows when temperatures are near 10°C. 18 – 25°C is an optimum range for growth; sturgeon will die when temperature is > 35°C. The optimum temperature for juvenile sturgeon is 22 – 25°C (Zhang, 1998).
V_2	Water Depth (Adults)	The Chinese sturgeon is distributed in areas with 9.3 – 40 m water depth; 90% of individuals are distributed at depths from 11 – 30 m; 11 Chinese sturgeons detected in the Yanzhiba to Gulaobei reach were distributed at depths from 9 – 19 m (Wei, 2005).
V_3	Substrate (Adults)	Juvenile and adult Chinese sturgeons have similar substrate choices as with short-nose sturgeon in the US. Experiments show that juvenile short-nose sturgeon prefer habitat in sand-mud substrate or gravel substrate (Pottle and Dadswell, 1979). Chinese sturgeon prefer to cruise along river channels with deep trenches and sandy dunes, and are fond of resting in pools, backwaters, and places varied terrain (Guo and Lian, 2001).
V_4	Water Temperature (Spawning)	The spawning temperature for sturgeon is 17.0 – 20.0°C; spawning will stop when temperature < l6.5°C (Hu et al., 1992). The average temperature in the reaches downstream of the Gezhouba Dam during the sturgeon spawning period is 15.8 – 20.7°C. About 79.31% of fish are spread in the range of 17.5 – 19.5 °C; the average temperature of the original spawning sites in the upper reaches of the Yangtze River is 17.0 – 20.2°C. Therefore, the suitable spawning temperature for Chinese sturgeon is 17.0 – 20.0°C (Wei, 2003). Spawning occurs when temperature is 15.3– 20.5°C; the suitable range is 17.0 – 20.0°C, and the optimum in 18.0 – 20.0°C (Yang et al., 2007).

Variables	Eco-factors	Results of Previous Research
V_5	Water Depth (Spawning)	More than 20 years' of monitoring indicates that the length of new spawning sites is about 30 km from the tail water area of Gezhouba Dam to Gulabei, with 10 – 15 m water depth (Guo and Lian, 2001). The "stable spawning site of Chinese sturgeon" determined by Deng et al. (1991) has a water depth in a range from 4 – 10 m.
V_6	Substrate (Spawning and hatching)	Gravel and pebbles are present in Chinese sturgeon spawning sites of (Li, 1999). The substrate of new spawning sites are composed of sand, grave with sand, gravel and stone, and gradually coarsen from left to right bank (Hu et al., 1992). The substrate of the original centralized spawning sites of Chinese sturgeon was mainly composed of stones and gravels (Xing, 2003).
V_7	Water Temperature (Hatching)	The suitable temperature for hatching is 16 – 22°C, the optimum is 17 – 21°C. The hatching rate decreases when at temperature < 16°C; deformity rate increases at temperature > 23°C. The temperature should be stable when zoosperms are hatching, abnormal fetation or death will occur with even small fluctuations in temperature of 3 – 5°C (NERCITA, 2004b). Water temperature for cultivating fries should be between 12 – 29°C; the most suitable temperature is 16 – 24°C (Wang et al., 2002).
V_8	Flow Velocity (Spawning)	Sturgeon prefer spawning areas with flow velocity of 0.08–0.14 m/s at the bottom, 0.43 – 0.58 m/s in the middle, and 1.15 – 1.70 m/s at the surface (Li, 2001). The surface flow velocity at spawning areas is 1.1 – 1.7 m/s (Li, 1999). The flow velocity of spawning areas during spawning season ranges from 0.82 – 2.01 m/s; 57.69% of fish are distributed between 1.2 – 1.5 m/s. When spawning occurs during periods when water levels are falling, the daily fluctuation range of flow velocity is 0.82 – 1.86 m/s, with an average of 1.24 m/s. The daily maximum fluctuation range is 1.20 – 2.33 m/s, with an average of 1.56 m/s. When spawning activity occurs during periods when water levels are rising, the daily fluctuation range of flow velocity is 1.17 – 2.01 m/s, with an average of 1.55 m/s (Wei, 2005). According to 31 records from 1983 – 2000, the average flow velocity on spawning day was between 0.81 – 1.98 m/s, and 81% took place in the range of 1.00 – 1.66 m/s (Yang et al., 2007).
V_9	Suspended Sediment Concentration (Spawning)	The suspended sediment concentration in reaches downstream from the Gezhouba Dam is between 0.073 – 1.290 kg/m^3, with an average of 0.508 kg/m^3. About 66.67% of fish are distributed between 0.3 – 0.7 kg/m^3. When spawning activity occurs during periods of falling water level, the daily average suspended sediment concentration varies between 0.17 – 1.29 kg/m^3, with an average of 0.52 kg/m^3. When spawning activity occurs during periods of rising water levels, the daily average suspended sediment concentration varies between 0.41 – 1.02 kg/m^3, with an average of 0.61 kg/m^3 (Wei, 2005). The suitable range of suspended sediment concentration for Chinese sturgeon is 0.10 – 1.32 kg/m^3. From 1983 – 2000, 15 of 31 spawning events were in the range of 0.2 – 0.3 kg/m^3 (Yang et al., 2007).

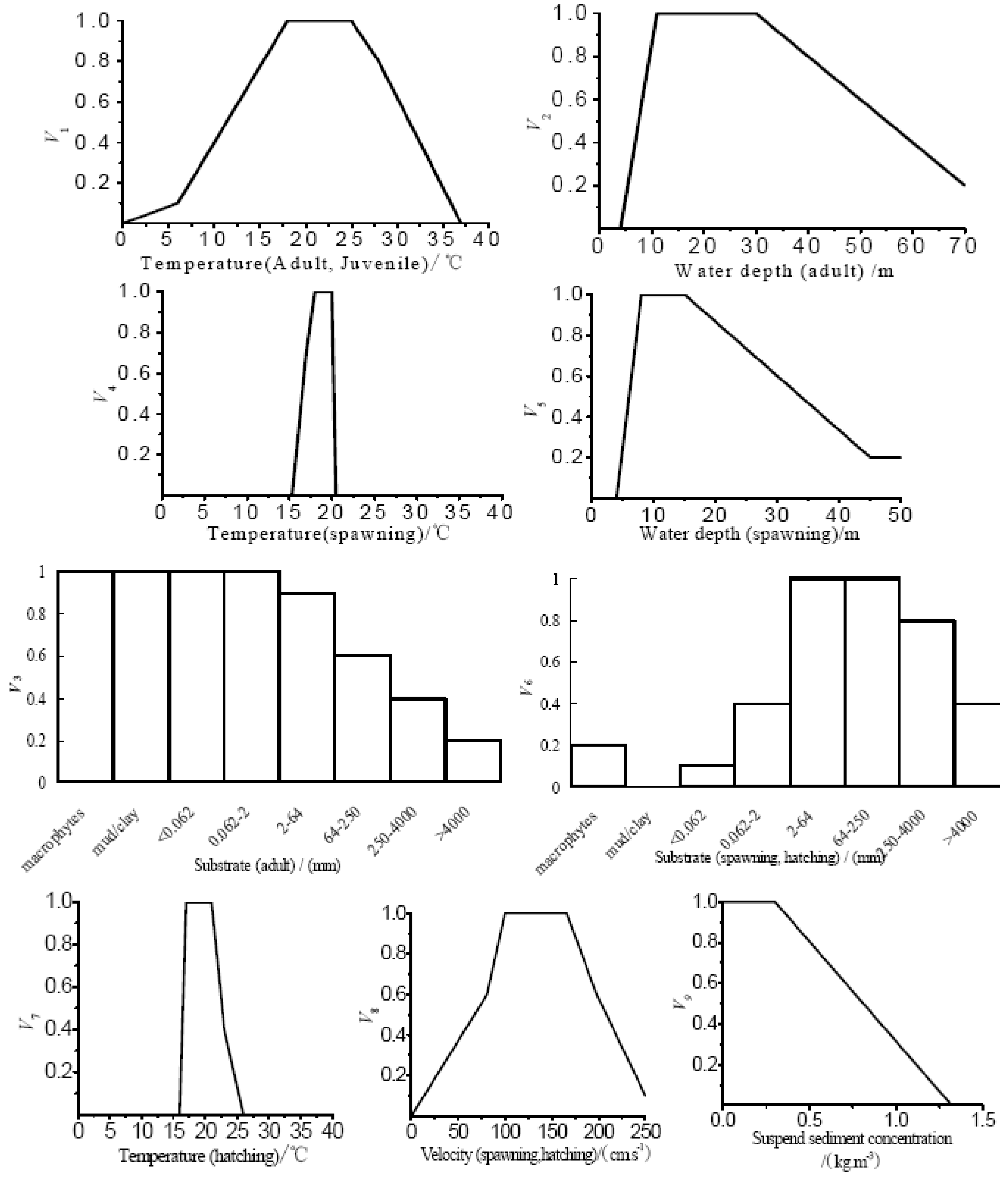

Figure 46. Suitability Index curves for habitat of Chinese sturgeon.

3.4.5. Vegetation-Hydroperiod Modeling

Vegetation-Hydroperiod Modeling is a very useful tool for habitat evaluation. Hydroperiod is defined as the depth, duration, and fequency of inundation and is a powerful determinant of what plants are likely to be found in various positions in the riparian zone, as shown in Figure 47. In most cases, the dominant factor that makes the riparian zone distinct from the surrounding uplands, and the most important gradient in structuring variation within the riparian zone, is site moisture conditions, or hydroperiod. Formalizing this relation as a vegetation-hydroperiod model can provide a powerful tool for analyzing existing distributions of riparian vegetation, casting forward or backward in time to alternative distributions, and designing new distributions.

Figure 47. Soil moisture conditions determine the plant communities in riparian areas of the Nile River in Sudan.

The suitability of site conditions for various species of plants can be described with the same conceptual approach used to model habitat suitability for animals. The basic logic of a vegetation-hydroperiod model is straightforward. It is possible to measure how wet a site is and, more importantly, to predict how wet a site will be. From this, it is possible to estimate what vegetation is likely to occur on the site.

The two basic elements of the vegetation-hydroperiod relation are the physical conditions of site moisture at various locations and the suitability of those sites for various plant species. In the simplest case of describing existing patterns, site moisture and vegetation can be directly measured at a number of locations. However, to use the vegetation-hydroperiod model to predict or design new situations, it is necessary to predict new site moisture conditions. The most useful vegetation-hydroperiod models have the following three components (FISRWG, 1997):

1) Characterization of the hydrology or pattern of stream flow- This can take the form of a specific sequence of flows, a summary of how often different flows occur, such as a flow duration or flood frequency curve, or a representative flow value, such as bankfull discharge or mean annual discharge.
2) A relation between streamflow and moisture conditions at sites in the riparian zone- This relation can be measured as the water surface elevation at a variety of discharges and summarized as a stage vs. discharge curve. It can also be calculated by a number of hydraulic models that relate water surface elevations to discharge, taking into account variables of channel geometry and roughness or resistance to flow. In some cases, differences in simple elevation above the channel bottom may serve as a reasonable approximation of differences in inundating discharge.

3) A relation between site moisture conditions and the actual or potential vegetation distribution- This relation expresses the suitability of a site for a plant species or cover type based on the moisture conditions at the site. It can be determined by sampling the distribution of vegetation at a variety of sites with known moisture conditions and then deriving probability distributions of the likelihood of finding a plant on a site given the moisture conditions at the site. General relations are also available from the literature for many species.

In altered or degraded stream systems, current moisture conditions in the riparian zone may be dramatically unsuitable for the current, historical, or desired riparian vegetation. Several conditions can be relatively easily identified by comparing the distribution of vegetation to the distribution of vegetation suitabilities.

The hydrology of the stream has been altered, for example, if stream flow has diminished by diversion or flood attenuation, sites in the riparian zone may be drier and no longer suitable for the historic vegetation or for current long-lived vegetation that was established under a previous hydrologic regime. The inundating discharges of plots in the riparian zone have been altered so that streamflow no longer has the same relation to site moisture conditions; for example, levees, channel modifications, and bank treatments may have either increased or decreased the discharge required to inundate plots in the riparian zone. The vegetation of the riparian zone has been directly altered, for example, by clearing or planting so that the vegetation on plots no longer corresponds to the natural vegetation for which the plots are suitable. Temporal variability is a particularly important characteristic of many stream ecosystems. Regular seasonal differences in biological requirements are examples of temporal variability that are often incorporated into biological analyses based on habitat suitability and time series simulations. The need for episodic extreme events is easy to ignore because these are as widely perceived as destructive both to biota and constructed river features. In reality, however, these extreme events seem to be essential to physical channel maintenance and to the long-term suitability of the riverine ecosystem for disturbance-dependent species. Cottonwood in riparian systems in the western U.S. is one well understood case of a disturbance-dependent species. Cottonwood regeneration from seed is generally restricted to bare, moist sites. Creating these sites depends heavily on channel movement (meandering, narrowing, avulsion) or new flood deposits at high elevations. In some riparian systems, channel movement and sediment deposition on flood plains tend to occur infrequently in association with floods. The same events are also responsible for destroying stands of trees. Thus, maintaining good conditions for existing stands, or fixing the location of a stream's banks with structural measures, tends to reduce the regeneration potential and the longterm importance of this disturbance-dependent species in the system as a whole.

There is a large body of information on the flooding tolerances of various plant species. Summaries of this literature include Whitlow and Harris (1979) and the multivolume Impact of Water Level Changes on Woody Riparian and Wetland Communities (Teskey and Hinckley, 1978; Walters et al., 1978; Lee and Hinckley, 1982; Chapman et al., 1982). This type of information can be coupled to site moisture conditions predicted by applying discharge estimates or flood frequency analyses to the inundating discharges of sites in the riparian zone. The resulting relation can be used to describe the suitability of sites for various plant species, e.g., relatively floodprone sites will likely have relatively flood-tolerant plants. Inundating discharge is strongly related to relative elevation within the floodplain. Other things being equal (i.e., within a limited geographic area and with roughly equivalent hydrologic regimes), elevation relative to a representative water surface line, such as bankfull discharge or the stage at mean annual flow, can, thus, provide a reasonable surrogate for site moisture conditions. Locally determined vegetation suitability can then be used to determine the likely vegetation in various elevation zones.

4. RESTORATION STRATEGIES

"Leave it alone and let it heal itself." In some cases, the best solution to a river ecological problem might be to remove the stresses and "let it heal itself." Unfortunately, in many cases this process can take quite a long time. Therefore, the "leave it alone" concept is a difficult approach for people to accept (Gordon et al., 1992). Restoration of the impaired river ecosystem is necessary. According to the U.S. National Research Council (NRC, 1992), restoration should involve the return of a given ecosystem to a state approximating that in which it existed prior to disturbances.

4.1. Design of stream Restoration

Design of instream habitat restoration can be guided and fine tuned by assessing the quality and quantity of habitat provided by the proposed design. It should be noted, however, that the best approach to habitat recovery is to restore a fully functional, well-vegetated stream corridor within a well-managed watershed. Man-made structures are less sustainable and rarely result in a stable channel. Over the long term, design should rely on natural fluvial processes interacting with floodplain vegetation and associated woody debris to provide high-quality aquatic habitat. Structures have little effect on populations that are limited by factors other than physical habitat. Newbury and Gaboury (1993) and Garcia (1995) adapted the following procedures to restore instream habitat.

1. Select stream. Give priority to reaches with the greatest difference between actual (low) and potential (high) fish carrying capacity and with a high capacity for natural recovery processes.
2. Evaluate fish populations and their habitats. Give priority to reaches with habitats and species of special interest. Is this a biological, chemical, or physical problem? If a physical problem, do the following.
3. Diagnose physical habitat problems: *Drainage basin* -Trace watershed lines on topographical and geological maps to identify sample and rehabilitation basins. *Profiles* - Sketch main stem and tributary long profiles to identify discontinuities that might cause abrupt changes in stream characteristics (falls, former base levels, etc.). *Flow* - Prepare flow summary for rehabilitation reach using existing or nearby records if available (flood frequency, minimum flows, historical mass curve). *Channel geometry survey* - Select and survey sample reaches to establish the relation between channel geometry, drainage area, and bankfull channel-forming discharge. Quantify hydraulic parameters at design discharge. *Rehabilitation reach survey*- Survey rehabilitation reaches in sufficient detail to prepare channel cross section

profiles and construction drawings and to establish survey reference markers. *Preferred habitat* - Prepare a summary of habitat factors for biologically preferred reaches using regional references and surveys. Identify multiple limiting factors for the species and life stages of greatest concern. Where possible, undertake reach surveys in reference streams with proven populations to identify local flow conditions, substrate, refuges, etc.

4. Design a habitat improvement plan. Quantify the desired results in terms of hydraulic changes, habitat improvement, and population increases. Integrate selection and sizing of habilitation works with instream flow requirements. Select potential schemes and structures that will be reinforced by the existing stream dynamics and geometry. Test designs for minimum and maximum flows and set target flows for critical periods derived from the historical mass curve.
5. Implement planned measures. Arrange for on-site location and elevation surveys and provide advice for finishing details in the stream.
6. Monitor and evaluate results. Arrange for periodic surveys of the rehabilitated reach and reference reaches, to improve the design, as the channel ages.

Evidence suggests that traditional design criteria for widespread bank and bed stabilization measures (e. g., concrete grade control structures, homogeneous riprap) can be modified, with no functional loss, to better meet environmental objectives and improve habitat diversity. Weirs are generally more failure-prone than deflectors. Deflectors and random rocks are minimally effective in environments where higher flows do not produce sufficient local velocities to produce scour holes near structures. Random rocks (boulders) are especially susceptible to undermining and burial when placed in sand-bed channels, although all types of stone structures experience similar problems. Additional guidance for evaluating the general suitability of various fish habitat structures for a wide range of morphological stream types is provided by Rosgen (1996). Seehorn (1985) provides guidance for small streams in the eastern U.S. Nowadays numerous design web sites are available (White and Brynildson, 1967, Seehorn, 1985, Wesche, 1985, Orsborn et al., 1992, Orth and White, 1993, Flosi and Reynolds, 1994). The use of any of these guides should also consider the relative stability of the stream, including aggradation and incision trends, for final design. Hydraulic conditions at the design flow should provide the desired habitat; however, performance should also be evaluated at higher and lower flows. Barriers to movement, such as extremely shallow reaches or vertical drops not submerged at higher flows, should be avoided. If the conveyance of the channel is an issue, the effect of the proposed structures on stages at high flow should be investigated. Structures may be included in a standard backwater calculation model as contractions, low weirs, or increased flow resistance (Manning) coefficients, but the amount of increase is a matter of judgment or limited by National Flood Insurance Program ordinances. Scour holes should be included in the channel geometry downstream of weirs and dikes since a major portion of the head loss occurs in the scour hole. Hydraulic analysis should include estimation or computation of velocities or shear stresses to be experienced by the structure. If the hydraulic analysis indicates a shift in the stage-discharge relation, the sediment-rating curve of the restored reach also may change, leading to deposition or erosion. Although modeling analyses are usually not cost-effective for a habitat structure design effort, informal analyses based on assumed relations between velocity and sediment discharge at the bankfull discharge may be helpful in detecting potential problems. An effort should be made to predict the locations and magnitude of local scour and deposition. Areas projected to experience significant scour and deposition should be sites for visual monitoring after construction. Materials used for aquatic habitat structures include stone, fencing wire, posts, and felled trees.

Priority should be given to materials that occur on site under natural conditions. In some cases, it may be possible to salvage rocks or logs generated from construction of channels or other project features. Logs give long service if continuously submerged. Even logs not continuously wet can give several decades of service if chosen from decay-resistant species. Logs and timbers must be firmly fastened together with bolts or rebar and must be well anchored to banks and bed. Stone size should be selected based on design velocities or shear stress.

4.2. Instream Structures for Habitat Restoration

Artificial instream structures have been used to modify fish habitat conditions from about the mid 1930s in the U.S. Since that time the use of these structures has gradually increased worldwide. Such techniques have been increasingly used to rehabilitate habitat in rivers impacted by channel works or adversely affected by regulation (Brookes and Shields, 1996). It appears that the techniques were initially used to improve rivers, which had a recreational value for salmonids, but more recently have been extended to degraded rivers also containing non-salmonid species. Instream structures are just one of the more localized methods of improving habitat. Common types of instream structures include weirs, deflectors, random rocks, bank covers, substrate reinstatement, fish passage structures, and off-channel ponds and coves. Habitat structures have been used more along cold water streams. In 1995, 1,234 structures were evaluated according to their general effectiveness, the habitat quality associated with the given structure type, and actual use of the structures by fish (Bio West, 1995). The study concluded that instream habitat structures generally provided increased fish habitat, but 18 percent of the structures need maintenance. Where excessive sediment delivery occurs, structures have a brief lifespan and limited value in terms of habitat improvement. The typical structures are listed in Table 9.

Table 9. Structural techniques used for instream habitat improvement (after Brookes and Shields, 1996)

Type	Functions
Deflectors	To direct flow and eliminate accumulated sediment or to narrow a channel, thereby increasing the velocity and creating a scour pool with a corresponding downstream riffle.
Small weirs or sills	Diversify habitat by impounding a greater depth of flow above the structure and by increasing the velocity downstream to erode a scour pool
Substrate placement	Placement of new substrate to enhance the habitat for fish and macro invertebrates.
Devices providing direct cover	Fixed to the bed or banks of a channel to float and adjust their level with varying discharge

Species-Centered Restoration - Many angling organizations have used instream structures to improve habitat for maximum production of salmonids and other game fish. In the U.S. there has also been Federal involvement through agencies such as the U.S. Forest Service and the U.S. Fish and Wildlife Service since the 1930s. There are a considerable number of publications, which demonstrate how effective such projects can be for increasing fish numbers and biomass. However, sometimes this type of work has been detrimental to other species, such as beaver (NRC, 1992). Gore (1985) argues that restoration of macro invertebrate communities is essential since they often form a substantial portion of the food supply for fish. Maximizing habitat value for any single species is not the same as recreating the biotic structure and function of a stream, involving consideration of a number of species (NRC, 1992). A key characteristic of a productive stream is physical habitat diversity. It is essential that there are appropriate ranges of water depths, water velocities, and substrate types. Since cover is also important, consideration needs to be given to the riparian plant community (Hunt, 1988). Suitable depths and velocities are needed for spawning habitat, but if an appropriate substrate is absent then the habitat value is diminished for species, which spawn on the bed. Likewise, if substrate has been reinstated without consideration of holding pools for mature fish then the habitat value is again questionable. Many approaches have now been developed to quantify habitat value. The approaches are based on the concept that abundance of a particular species can be correlated with particular habitat requirements, For example, the U.S. Fish and Wildlife Service Physical Habitat Simulation System Model is useful for crude prediction and analysis of potential habitat improvement (Bovee and Milhous, 1978).

Deflectors - Perhaps the most widely used structures for habitat improvement are current deflectors. These function either to direct flow and eliminate accumulated sediment, or to narrow a channel, thereby locally increasing the velocity and creating a scour pool with a corresponding riffle downstream. Other effects include assisting the development of a meandering thalweg within a straightened channel, protecting stream banks from erosion, and encouraging the establishment of riparian vegetation through the formation of bars of sediment.

Deflectors are commonly angled in a downstream direction at approximately 45 degrees from the current (Wesche, 1985), although several different angles have been used depending on the local circumstances (Cooper and Wesche, 1976). Double wing deflectors, which consist of two current deflectors placed opposite each other at the same point in a reach, have also been used (Seehorn, 1985). Shape is another consideration and varies from single peninsular wing to a triangular wing (White and Brynildson, 1967). The latter shape has been successful in certain circumstances in regulating the tendency for erosion of the bed and bank behind the structure during high flows. The elevation of the water surface at low flow generally determines the structure height and it has been found that to avoid excessive damage to the structure itself during high flows the structure should not extend more than 0.15 to 0.30 m above the low flow elevation (Seehorn, 1985; Wesche, 1985). The distance that the deflector protrudes into the channel will vary from site to site and depends on the specific results desired. For example, in streams in the southeastern U.S., Seehorn (1985) found that to have any effect the channel would need to be narrowed to a width approximating the natural width. This width can be determined from adjacent or neighboring natural reaches with a similar slope, flow regime, and bed and bank materials. Deflectors are very effective at manipulating flow and creating the diversity of habitat required for fish and other biota. Figure 48. shows a spur dike extension on the Lijiang River, to promote the formation of low velocity habitats. Patterns of scour and deposition are evident. Such structures can be built in series, alternating from one bank to the other, to create a meandering thalweg. Several authors have recommended a spacing of between five and seven channel widths, corresponding to the natural pool-riffle spacing of some natural streams (White, 1975; Everhart et al., 1975). This helps to formalize the low flow sinuous channel within an wide flood channel (Brookes, 1995). Many of these structures demonstrated marked improvement in fish population and the benthic macroinvertebrates. However, rarely have studies adopted a more integrated approach of objectively assessing the potential hydraulic and geomorphologic impacts at the outset, and most have been aimed at the restoration of single species. An understanding of how deflectors typically function to provide habitat may aid in the selection of building material and design. Whilst deflectors by design, create and maintain pool-riffle sequences, they also provide zones of higher velocities. These areas are critical to some species of fish and macro invertebrates. Swiftly moving water transports food and oxygen into a river reach. To exploit these resources fish must either swim against the current at a rather high cost of energy to maintain their position, or they must find sheltered areas close enough to the fast water to derive the benefits of higher dissolved oxygen and food availability. Although many organisms typical of high velocity reaches are physically adapted to maintain position or spatial orientation, most species require ambush locations out of the direct flow of water. In unaltered streams, organisms use velocity shelters provided by undercut banks, boulders or large woody debris, but if these natural habitats are absent in degraded reaches, current deflectors may provide them. Studies in large rivers and small streams have shown that stone deflectors (groins, spur dikes) used for erosion control provide aquatic habitat superior to stone blanket revetment, which is not as effective in creating the juxtaposition of high and low velocity zones (Shields et al., 1995)

Figure 48. Spur dikes on the Lijiang River in south China used to promote the formation of low velocity habitats.

Weirs - Weirs also increase habitat diversity through creation of pool-riffle characteristics. Weirs constructed of locally derived material are relatively low cost, and can be built from a variety of materials, including logs, boulders, stones, and gabions. Wesche (1985) covers various designs in detail. Weirs can be placed to break up the flow of the river and increase turbulence downstream perhaps scouring a pool, whist impounding water upstream. Figure 49. illustrates a weir placed on the Juma River in the suburbs of Beijing for creation and maintenance of a habitat with stable low current and high depth of water. In dry seasons the weir maintains a depth of water for fish and invertebrates to survive. Downstream from weirs sediment may be eroded to form a scour hole and the sediment may be transported for a distance and deposit to form a riffle-like feature. Weirs have been also used to impound flow to allow fish passage, to trap for gravel spawning moving along high gradient streams, to trap finer sediments, to aerate the water, and to slow the flow, enabling organic debris to fall out and enhance invertebrate production (Wesche, 1985). Such structures extend across the entire width of the channel, although some incorporate a notch to locally concentrate flow. This type of structure is perhaps most effective for producing a pool-riffle pattern in low energy streams. Habitat provided by weirs can be particularly valuable in channelized streams experiencing relatively high levels of erosion and deposition. Cooper and Knight (1987) compared fish catches from pools below weirs used as grade control structures and natural scour pools in unstable, channelized streams in Mississippi. U.S. Grade control pools produced greater catches by weight, more fish of harvestable size, and more stable length frequency distributions than natural scour holes. Cooper and Knight (1987) speculated that habitat created by grade control structures was more stable than natural scour holes, which undergo frequent cycles of filling and scouring. This stability resulted in more consistent successful spawning and recruitment and ultimately higher yields. Weirs have been used successfully in many countries to increase fish population within relatively short periods of time (Gard, 1961, McCall and Knox, 1978, Carling and Kloslewski, 1981, Shields et al., 1995). Artificial pools created by weirs in New Mexico, U.S. were up to 70% greater by volume than natural pools and held 50% more trout with twice the biomass.

Figure 49. A weir placed on the Juma River in the suburbs of Beijing for creation and maintenance of a habitat with stable low current and high depth of water. In dry seasons the weir maintains a depth of water for fish and invertebrates to survive.

There are also many examples of failures (Wesche, 1985). The structure may fail in high-energy rivers, particularly at high flow and they tend to be unsuccessful in enhancing habitat where there is an excessively high sediment load (Keown, 1981). Structure failure may not necessarily equate to failed habitat improvement. For example, rock scattered from a failed deflector may provide substrate for invertebrates, resting areas, or ambush sites for stream fish. Other reasons for failure to enhance biological populations include the blockage of fish passage (Johnson, 1971) and the lack of an adequate food supply (Rockett, 1979). *Modification of Substrate* - Boulders have been placed in channels to provide cover for fish, to improve the pool-riffle characteristics, to provide additional habitat for rearing fish or to protect banks from erosion. Randomly placed boulders can enhance fish habitat substantially (Knox, 1982, Lere, 1982). Diamond-shaped clusters of four boulders are often used. A field experiment was conducted by replacing the substrate with gravel, stones, sand and silt. Figure 50. shows the experiment in the Juma River in the suburbs of Beijing. The highest species richness was obtained with gravel substrate, proving that replacing substrate with gravel may effectively improve the aquatic ecology. In lowland streams where no endemic rock exists, large logs or wooden pilings may have ecological advantages over boulders. Woody debris is a key component of aquatic habitats in sand-bed rivers (Shields and Smith, 1992). Although woody structures do not last as long as rock, they provide a carbon source and may be more acceptable to organisms that have evolved to live on submerged woody debris. Wooden substrate may also be cheaper and more readily available. From an ecological point of view the placement of more natural bed sediments may speed recovery (Gore, 1985). Placement of artificial materials such as crushed limestone and quarry rejects may also improve the habitat for fish and for macro-invertebrates (Stuart, 1960). For example, on the Afon Gwyfai in Wales decolonization of invertebrates on a reinstated gravel bed was a gradual process, taking about a year (Brookes, 1988). The stability of reinstated gravel is a key issue. If the gravel is unstable, then the diversity and abundance of species will be less than for a stable bed. In a high-energy environment it may be necessary to install structures to retain gravels in situ (Claire, 1980). Also, where sediment loads are too high due to upstream channel modification or land use change then spawning gravels may become smothered. This is a particular problem in low-energy lowland streams.

Figure 50. Experiment of replacing substrate with gravel, stones, sand, and silt in the Juma River near Beijing.

Devices Providing Cover - Under natural circumstances undercut banks and overhanging vegetation are important habitat features referred to as “cover”. Fish utilize cover for shade and shelter. Figure 51. shows riparian trees and wood logs in the water that provide shade and shelter for aquatic wildlife in the Jiuzhaigou Creek in the upper Jialing River basin, which attract many fish in the river. Artificial devices may be fixed to the bed or banks to provide additional cover. They include log overhangs, overhanging platforms, felled trees, which have been anchored in place, and riprap (Claire, 1980). These have been demonstrated to be particularly effective at increasing the number of trout in a reach (White, 1975).

Artificial “fish attractors” (a cover device made of brush, bundles of old tires) have been used and extensively tested in North American reservoirs, but less attention has been paid to rivers and streams. Wilbur (1978) found that materials used to construct the attractor determined which species exploited the created habitat. Brush and attractors made from the branches of trees were reported to be somewhat more successful in attracting fish than attractors constructed of other materials. Anecdotal evidence indicates that the spacing and configuration of branches may play an important role in attracting fish, so some consideration of plant species is needed when brush piles or tree reefs are used as cover in streams.

Engineered Log Jams - Woody debris within a stream can often influence the instream channel structure by increasing the occurrence of pools and riffles. As a result, streams with woody debris typically have less erosion, slower routing of organic detritus (the main food source for aquatic invertebrates), and greater habitat diversity than straight, even-gradient streams with no debris. Woody debris also provides habitat cover for aquatic species and characteristics ideally suited for fish spawning.

Figure 51. Riparian trees and wood logs in the water provide shade and shelter for aquatic wildlife and attract many fish in the Jiuzhaigou Creek in upper Jialing River basin.

Reintroduction of woody debris, or log-jams, has been extensive, but limited understanding of woody debris stability has hampered many of these efforts. Engineered log jams can restore riverine habitat and in some situations can provide effective bank protection. Even in large alluvial channels that migrate at rates of 10 m/yr, jams can persist for centuries, creating a mosaic of stable sites that in turn host the large trees necessary to initiate stable jams (Abbe et al., 1997). Engineered log jams are designed to emulate natural jams and can meet management or restoration objectives such as habitat restoration and bank protection.

After learning about the uncertainty and potential risks of creating man-made log jams, landowners near Packwood, Washington, U.S. decided the potential environmental, economic, and aesthetic benefits outweighed the risks. An experimental project consisting of three engineered log jams was implemented to control severe erosion along 420 m of the upper Cowlitz River. Five weeks after constructing the log jams, the project experienced a 20-year return period flood (850 m^3/s). The engineered log jam remained intact and met design objectives by transforming an eroding shoreline into a local depositional environment (accreting shoreline). Approximately 93 tons of woody debris that was in transport during the flood was trapped by the jams, alleviating downstream hazards and enhancing structure stability (Abbe et al., 1997). Landowners have been delighted by the experiment.

4.3. Stream Restoration

In some cases, it might be desirable to divert a straightened stream into a meandering alignment for restoration purposes. For cases where the designed channel will carry only a small amount of bed material load, bed slope and channel dimensions may be selected to carry the design discharge at a velocity that will be great enough to prevent suspended sediment deposition and small enough to prevent erosion of the bed. Increasing the sinuosity of a stream may create better habitats for faunal communities. Meanders can then be created with length ranges from 4 to 9 times the channel widths. Meanders should not be uniform. For instance, the incised, straightened channel of the River Backwater in the U.K. was restored to a meandering form by excavating a new low-level floodplain about 50 to 65 feet wide containing a sinuous channel about 16 feet wide and 3 feet deep (Hey, 1995). Preliminary calculations indicated that the bed of the channel was only slightly mobile at bank-full discharge, and sediment loads were low.

For small rivers restoration practices also involve change of the channel dimensions, in which the average values for width and depth of the channel were determined in the design. These determinations are based on the imposed water and sediment discharge, bed sediment size, bank vegetation, resistance, and average bed slope. However, both width and depth may be constrained by site factors, which the designer must consider once stability criteria are met. Perhaps the simplest approach to selecting channel width and depth is to use dimensions from stable reaches elsewhere in the watershed or from similar reaches in the region. A reference reach is a reach with a desired biological condition, which will be used as a target to strive for when comparing various restoration options. A reference reach used for stable channel design should be evaluated to make sure that it is stable and has a desirable ecological condition. In addition, it must be similar to the desired project reach in hydrology, sediment load, and bed and bank material. Often a stable reach upstream of or downstream from the reach to be restored is selected as the reference reach.

Stabilization of the bed and banks – Stabilization of the bed and banks is often required for eco-system restoration. Plants may be established on upper bank and floodplain areas using traditional techniques for seeding or by planting bare-root and container-grown plants.

However, these approaches provide little initial resistance to flows, and plantings may be destroyed if subjected to high water before they are fully established. Cuttings, pole plantings, and live stakes taken from species that sprout readily (e.g., willows) are more resistant to erosion and can be used lower on the bank (Figure 52).

In addition, cuttings and pole plantings can provide immediate moderation of flow velocities if planted at high densities. Often, they can be placed deep enough to maintain contact with adequate soil moisture levels, thereby eliminating the need for irrigation. The reliable sprouting properties, rapid growth, and general availability of cuttings of willows and other pioneer species makes them particularly appropriate for use in bank revegetation projects, and they are used in most of the integrated bank protection approaches.

Geotextile Systems - Geotextiles have been used for erosion control on road embankments and other upland settings, or with plants placed through slits in the fabric. In self-sustaining stream bank applications, only natural, biodegradable materials should be used, such as jute or coconut fiber (Johnson, 1994).

The typical stream-bank use for these materials is in the construction of vegetated geo-grids, which are similar to brush layers except that the fill soils between the layers of cuttings are encased in fabric, allowing the bank to be constructed of successive "lifts" of soil, alternating with brush layers. This approach allows reconstruction of a bank and provides considerable erosion resistance.

Natural fibers are also used in "Fiber-Schines," which are sold specifically for stream-bank applications. These are cylindrical fiber bundles that can be staked to a bank with cuttings or rooted plants inserted through or into the material. Vegetated plastic geo-grids and other non-degradable materials can be used where geo-technical problems require drainage or additional strength.

Tree revetments – Tree revetments are made from whole tree trunks laid parallel to the bank, and cabled to piles or deadman anchors. Eastern red cedar (Juniperus Virginian) and other coniferous trees are used on small streams, where their springy branches provide interference to flow and trap sediment.

The principal objective to these systems is the use of large amounts of cable and the potential for trees to be dislodged and cause downstream damage. Some projects have successfully used large trees in conjunction with stone to provide bank protection as well as improved aquatic habitat cut into the bank, such that the root wads extend beyond the bank face at the toe (Figure 53).

The logs are overlapped and/or braced with stone to ensure stability, and the protruding rootwads effectively reduce flow velocities at the toe and over a range of flow elevations (Figure 54).

A major advantage of this approach is that it reestablishes one of the natural functions of large woody debris in streams by creating a dynamic near-bank environment that traps organic material and provides colonization substrates for invertebrates and refuge habitat for fish. The logs eventually rot, resulting in a more natural bank. The revetment stabilizes the bank until woody vegetation has matured, at which time the channel can return to a more natural pattern.

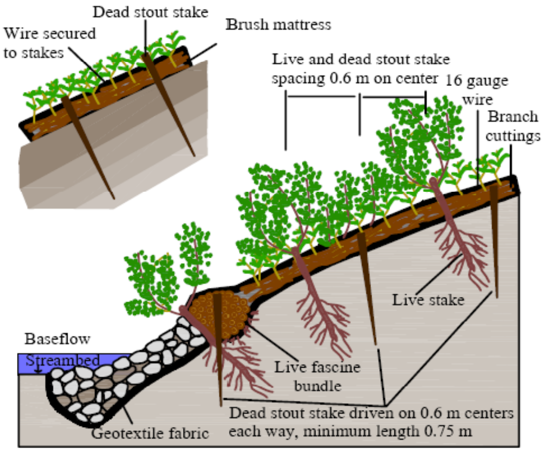

Figure 52. The reliable sprouting properties, rapid growth, and general availability of cuttings of willows and other pioneer species re-vegetate the banks quickly (after FISRWG, 1997).

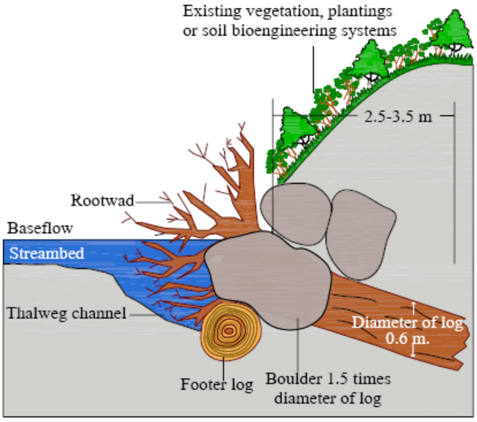

Figure 53. Large trees and stone are used to provide bank protection (after FISRWG, 1997).

Figure 54. The protruding rootwads effectively reduce flow velocities at the toe (after FISRWG, 1997).

4.4. Artificial Wetlands

Artificial wetlands have been created for ecological restoration. Wetland-loss has become a problem in many river systems as a result of sediment reduction, railway and highway construction, and drainage system development. The floral communities and faunal communities in the wetlands suffer from development and wetland-loss. To restore the wetland ecology artificial wetlands, for instance green tree reservoirs, are applied. Greentree reservoirs are shallow, forested flood-plain impoundments usually created by building low levees and installing outlet structures. They are usually flooded in early fall and drained during late March to mid-April. Draining prevents damage to over-story hardwoods (Rudolph and Hunter, 1964). Most existing green tree reservoirs are in the southwestern U.S. The green tree reservoirs provide habitat for many animal species. The flooding of green tree reservoirs differs from the natural flood regime. Green tree reservoirs are typically flooded earlier and at depths greater than would normally occur under natural conditions. Over time, modifications of natural flood conditions can result in vegetation changes, lack of regeneration, decreased mast production, tree mortality, and disease. Proper management of Green tree reservoirs requires knowledge of the local system-especially the natural flood regime and sediment transport and deposition. Figure 55. shows an artificial wetland in the suburbs of Seoul, South Korea. Many birds have colonized the new habitat just a few years after the artificial wetland was created. In the meantime the wetland also attracts tourists. Is it certain that the species richness, number density, and other biodiversity indices of benthic invertebrates and fish will increase if river-linked riparian lakes or wetlands are created? A comparative study was performed to answer these questions with samples taken from a river-lined lake - Zengjiang Bay, a low flow area of the Zengjiang River channel at Zhengguo, and an isolated riparian lake - Xizhijiang Oxbow Lake in the East River basin (Wang et al., 2008a). Figure 56. shows the geographical locations of Zengjiang Bay, Zhengguo, and Xizhijiang Oxbow Lake. The Zengjiang River is a tributary of the lower reaches of the East River. Substrate in the main channel of the Zengjiang River consists of sand. No benthic invertebrates were found in the sample from the river at Zhengguo.

Figure 55. An artificial wetland in the suburbs of Seoul South Korea.

Zengjiang Bay is like a riparian lake with a 100 m wide outlet connecting the river. The river carries fine suspended sediment into the bay which is deposited there. A mud layer covers most of the bay, and some hydrophytes have colonized parts of the bay. The flow velocity and water depth in the bay vary in the range of 0–0.5 m/s and 0-3 m, respectively. Zengjiang Bay provides multiple habitats for benthic invertebrates. The species richness of samples taken from the bay is 31, and the abundance of individual invertebrates is 343 ind/m^2. The calculated Shannon-Weaver index and bio-community index are H = 2.58 and B = 15.05, both the highest value in the East River. There are many fish species in the bay. Table 10. lists the species from the three sites, which shows that the river-lined riparian wetland has much higher species richness than those from the river and isolated riparian lakes. The Xizhijiang Oxbow Lake was a section of the Xizhijiang River Channel, which flows into the East River at Huizhou, but became an oxbow lake since an artificial cutoff in the 1980s. It is separated from the Xizhijiang River by a highway, as shown in Figure 56.(b). Part of the lake has been converted into a fishpond.

Because of the separation there is almost no flow velocity in the lake, while the separation also cut off the fine sediment supply to the lake. The substrate consists mainly of sand, which has remained unchanged for 20 years since the cut off. There is no mud layer in the lake. Analysis of samples from the lake indicates that the species richness is only 3 and the number density is 22 ind/m^2. The Shannon-Weaver index and biocommunity index are H = 0.89 and B= 2.76, which are much lower than those for Zengjiang Bay. To develop the oxbow lake into a good habitat for benthic invertebrates and fish, the lake should again be connected with the river, allowing fine sediment to be carried into the lake and a mud layer to form. The connection will also increase the flow velocity and exchange of lake water with river water.

In the lower East River if riparian waters are created similar to Zengjiang Bay the ecology in the river may be greatly improved. One example is to reconnect the Xizhijiang Oxbow Lake with the river. The lake water may exchange with the river water and fish and invertebrates may spend a part of their life cycle in the lake and other parts of their life cycle in the river.

Table 10. Species richness and abundance of macro-invertebrates from river-linked riparian lake Zengjian Bay, the main stem channel of the Zengjiang River at Zhengguo and a separated riparian lake – the Xizhijian Oxbow Lake

Zengjiang Bay	*Corbiculidae C.fluminea* (113); Chironomidae (four species 44); Elmidae, *Stenelmis* (25); *Ceratopogonidae Bezzia* (25); Corixidae (21); *Limnodrilus* (23); *Semisulcospira* (20); Libellulidae (14); Ephemeridae (11); *Bellamya Purificata* (8); Macromiidae (6); *Bellamya* sp_1 (5); *Branchiura* (4); *Coenagrionidae Pseudagrion* (4); Gomphidae, *Trigomphus* (3); Ampullariidae (2); Psephenidae (2); Hydrophilidae *Hydrobius* (2); Tabanidae (2); *Lepidoptera* (1); *Acariformes* (1); Gomphidae, *Sinictinogomphus* (1) ; Palaemonidae (1); Tricladida (1); Baetidae (1); Heptageniidae (1); *Parafossarulus* (1); Elmidae (1)
Zhengguo	0
Xizhijiang Oxbow Lake	*Palaemonidae (13); Chironomidae (7); Hydrophilidae, Laccobius (2)*

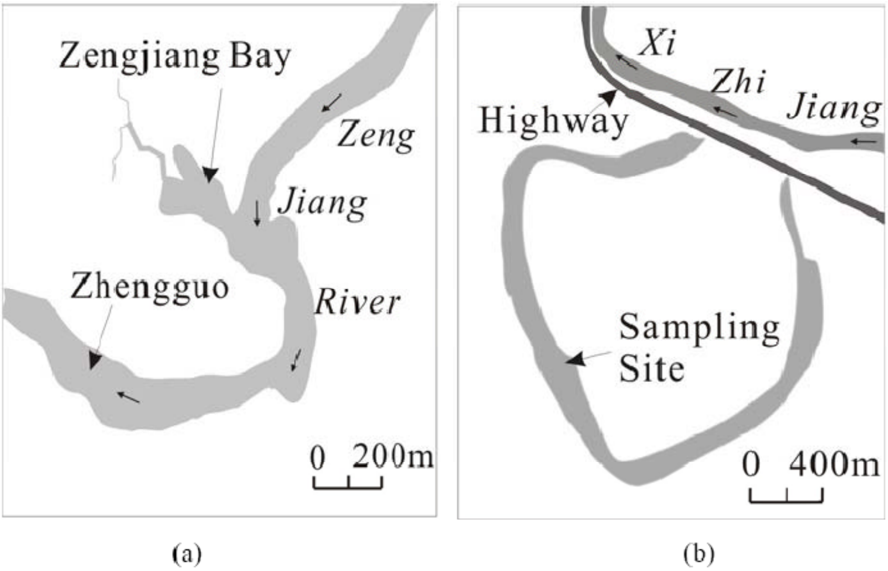

Figure 56. Location and shape of (a) Zengjiang Bay and (b) Xizhijiang Oxbow Lake.

4.5. Riparian Vegetation Restoration and Food Patches

Vegetation is a fundamental controlling factor for river eco-functions. Habitat, conduit, filter/barrier, and source/sink functions are all critically tied to the vegetative biomass amount, quality, and condition. Restoration designs should protect existing native vegetation and restore vegetative structure to result in a contiguous and connected stream corridor. Numerous shrubs and trees have been evaluated as restoration candidates, including willows (Svejcar et al., 1992, Anderson et al., 1978); alder, serviceberry, oceanspray, and vine maple (Flessner et al., 1992); Sitka and thin leaf alder (Java and Everett, 1992); paloverde and honey mesquite (Anderson et al., 1978); and many others. Selection of vegetative species may be based on the desire to provide habitat for a particular species of interest. The current trend in restoration, however, is to apply a multi-species or ecosystem approach.

The large-scale restoration of forest in the U.S. was undertaken by the Tennessee Valley Authority in conjunction with reservoir construction projects in the southern U.S. during the 1940s. Roads and railways were relocated outside the influence of the maximum pool elevation, but they were placed on embankments. The Tennessee Valley Authority was concerned that the roads and railways would be subject to wave erosion during periods of extreme high water. To reduce that possibility, agricultural fields between the reservoir and the embankments were planted with trees. At Kentucky Reservoir, approximately 4 km^2 of trees were planted, mostly on hydric soils adjacent to tributaries of the Tennessee River. Because the purpose of the planting was erosion control, little thought was given to recreating natural patterns of plant community composition and structure. Trees were evenly spaced in rows, and planted species were apparently chosen for maximum flood tolerance. As a result, the studied stands had an initial composition dominated by bald cypress, green ash, red maple, and water tolerant species, but they did not originally contain many of the other common bottomland forest species, such as oaks.

Figure 57. Reforestation of the desert in the middle reaches of the Yellow River.

In the middle reaches of the Yellow River to the north part of the Loess Plateau in China, a desert named the Maousu Desert extends over thousands of square kilometers. Very high rates of erosion occurred in the area, which impaired the terrestrial vegetation and caused detriment to al the river ecology. Great efforts have been made for erosion control in this area. Figure 57. shows the reforestation of the desert land. As a result of reforestation the sediment yield has greatly reduced, a plant community has developed, and some animals find their habitats in the area. Finally, the river ecology has improved.

Plant Community Restoration

Non-native vegetation can prevent establishment of desirable native species or become an unwanted permanent component of vegetation. For example, kudzu can kill forest species planted on out pasture grasses and weeds. Restoration work should restore natural patterns of plant community distribution by comparing with a reference plant community (Brinson et al., 1981,Wharton et al., 1982). Large-scale restoration work sometimes includes panting of understory species, particularly if they are required to meet specific objectives such as providing essential components of endangered species habitat. However, it is often difficult to establish understory species, which typically are not tolerant to full sun, if the restoration area is open. Where understory species are unlikely to establish themselves for many years, they can be introduced in adjacent forested sites, or planted after the initial tree growth to create appropriate understory conditions. Understory species seeds are commonly broadcast by hydroseeding, requiring a special tank truck with a pump and nozzle for spraying the mixture of seeds, fertilizer, binder, and water (Figure 58). A wider range of seed species can be planted more effectively with a seed drill towed behind a tractor (Haferkamp et al., 1985). Where access is limited, hand planting or aerial spreading of seeds might be feasible. In the past, stream corridor planting programs often included nonnative species selected for their rapid growth rates, soil binding characteristics, ability to produce abundant fruits for wildlife, or other perceived advantages over native species.

Figure 58. Hydroseeding of a streambank for restoration of understory riparian vegetation. Special tank trucks carrying seed, water, and fertilizer are used in the revegetation efforts (after FISRWG, 1997).

These actions sometimes have unintended consequences and often prove to be extremely detrimental (Olson and Knopf, 1986). As a result, many local agencies discourage or prohibit planting of nonnative species within wetlands or prohibit planting streamside buffers. It may be feasible in some cases to focus restoration actions on encouraging the success of local seed-fall to ensure that locally adapted populations of vegetation are maintained on the site (Friedman et al., 1995).

Nest Structures

Loss of riparian or terrestrial habitat in stream corridors has resulted in the decline of many species of birds and mammals that use associated trees and tree cavities for nesting or roosting. The most important limiting factor for cavity-nesting birds is usually the availability of nesting substrate (von Haartman, 1957), generally in the form of snags or dead limbs in live trees (Sedgwick and Knopf, 1986). Snags for nest structures can be created using explosives, girdling, or topping of trees. Artificial nest structures can compensate for a lack of natural sites in otherwise suitable habitat since many species of birds will readily use nest boxes or other artificial structures. For example, along the Mississippi River in Illinois and Wisconsin, U.S., where nest trees have become scarce, artificial nest structures have been erected and constructed for double-crested cormorants using utility poles (Yoakum et al., 1980). In many cases, increases in breeding bird density have resulted from providing such structures (Strange et al., 1971, Brush, 1983). Artificial nest structures can also improve nestling survival (Cowan, 1959).

Nest structures must be properly designed and placed, meeting the biological needs of the target species (FISRWG, 1997). They should also be durable, predator-proof, and economical to build. Design specifications for nest boxes include hole diameter and shape, internal box volume, distance from the floor of the box to the opening, type of material used, whether an internal "ladder" is necessary, height of placement, and habitat type in which to place the box. Other types of nest structures include nest platforms for waterfowl and raptors; and tire nests for squirrels. Specifications for nest structures for riparian and wetland nesting species (including numerous Picids, passerines, waterfowl, and raptors) can be found in many sources including Yoakum et al. (1980), Kalmbach et al. (1969), and various state wildlife agency and conservation publications.

Food Patches

Food patch planting is often expensive and not always predictable, but it can be carried out in wetlands or riparian systems mostly for the benefit of waterfowl. Environmental requirements of the food plants native to the area, proper time of year for introduction, management of water levels, and soil types must all be taken into consideration. Some of the more important food plants in wetlands include pondweed (*Potamogeton* spp.), smartweed (*Polyhonum* spp.), duckweeds (*Lemna* spp.), coontail, alkali bulrush (*Scirpus paludosus*), and various grasses. Two commonly planted native species include wild rice (*Zizania*) and wild millet. Details on these species can be found in Yoakum et al. (1980).

4.6. Restoration of Impounded and Channelized Streams

Damming of rivers may greatly change the hydrology, morphology and ecological conditions. Reservoir operation alters the flow of water, sediment, organic matter, and nutrients, resulting in both direct physical and indirect biological effects in tailwaters and downstream riparian and floodplain areas. Stream corridors below dams can be partially restored by modifying operation and management approaches.

Figure 59. Fish ladder on the Bonneville Dam on the Columbia River. More than 1 million salmon swim through the fish ladder to pass the dam to upstream spawning grounds every year.

The modification of operation approaches, where possible, in combination with the application of properly designed and applied best management practices, can reduce the impacts caused by dams on downstream riparian and floodplain habitats. Partial restoration of stream corridors below dams can be achieved by designing operation procedures that mimic the natural hydrograph. Modifications include scheduling releases, and making seasonal adjustments in pool levels and in the timing and variation of the rates of drawdowns (USEPA, 1993).

Adequate fish passage around dams, diversions, and other obstructions may be a critically important component of restoring healthy fish populations to previously degraded rivers and streams. However, designing installing, and operating a fish passage facility and the flows necessary for operation are generally site specific. Figure 59. shows the fish ladder on the Bonneville Dam on the Columbia River, U.S. More than 1 million salmon swim through the fish ladder to pass the dam to upstream spawning grounds every year.

Channelization and diversions represent forms of hydrologic modification commonly associated with most principal land uses, and their effects should be considered in all restoration efforts. In some cases, redesign of channel modifications to restore preexisting ecological characteristics is needed. All of the urban channels in Beijing have been channelized with concrete bed and banks. Riparian plant community and benthic invertebrates have lost their habitat and the water quality has become worse. It is planned to replace the concrete bed and banks with soil and stones, providing habitat for invertebrates. It is estimated that a high water quality may be maintained with only half of the water consumption.

Modifications of existing projects, including operation and maintenance or management, can improve some negative effects without changing the existing benefits or creating additional problems. Levees may be set back from the stream channel to better define the stream corridor and reestablish some or all of the natural floodplain functions. Setback levees can be constructed to allow for overbank flooding, which provides surface water contact with streamside areas such as floodplains and wetlands.

Instream modifications such as uniform cross sections or armoring associated with channelization or flow diversions may be removed, and design and placement of meanders can be used to reestablish more natural channel characteristics. In many cases, however, existing land uses might limit or prevent the removal of existing channel or floodplain modifications.

4.7. Restoration of Ecosystems Disturbed by Other Stresses

Exotic Species - Restoration of ecosystems disturbed by exotic species is difficult. Some land uses have actually introduced exotics that have become uncontrolled, while others have merely created an opportunity for such exotics to spread. Again, control of exotic species has some common aspects across land uses, but design approaches are different for each land use. Control of exotics in some situations can be extremely difficult and may be impractical if large areas or well-established populations are involved. Use of herbicides may be tightly regulated or precluded in many wetland and streamside environments, and for some exotic species there are no effective control measures that can be easily implemented over large areas (Rieger and Kreager 1990). Where aggressive exotics are present one should avoid unnecessary soil disturbance or disruption of intact native vegetation, and newly established populations of exotics should be eradicated.

Controlling exotics can be important because of potential competition with established native vegetation, colonized vegetation, and artificially planted vegetation, in restoration work. Exotics compete for moisture, sunlight, and space and can adversely influence establishment rates of new plantings. To improve the effectiveness of revegetation work, exotic vegetation should be cleared prior to planting; nonnative growth must also be controlled after planting. One must also understand the physical characteristics of the native vegetation for successful establishment, For example, native vegetation in the midwestern U.S. has learned to grow under conditions of low nutrient supply, and, thus, use of fertilizers often just promotes weed growth (Neal O'Reilly, Hey and Associates, personal commun). General techniques for control of exotics and weeds are mechanical (e.g., scalping or tilling), chemical (herbicides), and fire.

Agriculture - In agricultural areas when terraces and a waterway are installed in the nearby cropland, the scene depicts an ecologically deprived landscape. Nutrient and water flow, sediment trapping during floods, water storage, movement of flora and fauna, species diversity, interior habitat conditions, and provision of organic materials to aquatic communities are just a few of the functional conditions affected by these structural attributes.

Restoration design should establish functional connections within and external to stream corridors, landscape elements such as remnant patches of riparian vegetation, prairie, or forest exhibiting diverse or unique vegetative communities; productive land that can support ecological functions; reserve or abandoned land; associated wetlands or meadows; systems; ecologically innovative residential areas; neighboring springs and stream systems; ecologically innovative residential areas; and fauna (field borders, windbreaks, waterways, grassed terraces, etc.) offer opportunities to establish these connections. An edge (transition zone) that gradually changes from one land use into another will soften environmental gradients and minimize disturbance.

Urbanization often is the strongest disturbance to river ecosystems. The development of residence areas may severely impair the riparian vegetation and aquatic biocommunities. Seven restoration tools can be applied to help restore urban streams (FISRWG, 1997; Schueler, 1987). The best results are usually obtained when the following tools are applied together.

Tool 1: Partially restore the predevelopment hydrological regime. The primary objective is to reduce the frequency of bankfull flows by constructing upstream stormwater detention ponds that capture and detain increased storm water runoff for up to 24 hours before release (i.e., extended detention).

Tool 2: Reduce urban pollutant pulses. A second need in urban stream restoration is to reduce concentrations of nutrients, bacteria, and toxics in the stream, as well as trapping excess sediment loads. Generally, three tools can be applied to reduce pollutant inputs to an urban stream: storm water retrofit ponds or wetlands, watershed pollution prevention programs, and the elimination of illicit or illegal sanitary connections to the storm sewer network.

Tool 3: Stabilize channel morphology. Over time, urban stream channels enlarge resulting in severe bank and bed erosion. Therefore, it is important to stabilize the channel, and if possible, restore equilibrium channel geometry. In addition, it is also useful to provide undercuts or overhead cover to improve fish habitat. Depending on the stream order, watershed impervious cover, and the height and angle of eroded banks, a series of different tools can be applied to stabilize the channel, and prevent further erosion. Bank stabilization measures include imbricated riprap, brush bundles, and soil bioengineering methods, such as willow stakes and bio-logs and rootwads.

Tool 4: Restore instream habitat structure. Most urban streams have poor instream habitat structure, often typified by indistinct and shallow low flow channels within a much larger and unstable storm channel. The goal is to restore instream habitat structure that has been blown out by erosive floods. Key restoration elements include the creation of pools and riffles, deepening of the low flow channels, and the provision of greater structural complexity across the streambed. Typical tools include the installation of log check-dams, stone wing deflectors and boulder clusters along the stream channel.

Tool 5: Reestablish vegetative banks and riparian cover. Vegetative banks and riparian cover are essential components of the urban stream ecosystem. They stabilize banks, provide woody debris and detritus, and shade the stream. Therefore, the fifth tool involves reestablishing the vegetative banks and riparian cover plant community along the stream network. Figure 60. shows an urban stream in Beijing where layer of soil is placed on the hardened channel bed and banks, thus a vegetation bank and aquatic high plants have developed.

Tool 6: Protect critical stream substrates. A stable, well sorted streambed is often a critical requirement for fish spawning and secondary production by aquatic insects. The bed of urban streams, however, is often highly unstable and clogged by fine sediment deposits. It is often necessary to apply tools to restore the quality of stream substrates at points along the stream channel. Often, the energy of urban storm water can be used to create cleaner substrates - through the use of tools such as double wing deflectors and flow concentrators. If thick deposits of sediment have accumulated on the bed, mechanical sediment removal may be needed.

Figure 60. A layer of soil is placed on the hardened channel bed and banks on an urban stream in Beijing, and a vegetation bank and aquatic high plants have developed.

Tool 7: Allow for recolonization of the stream community. It may be difficult to reestablish the fish community in an urban stream if downstream fish barriers prevent natural recolonization. Thus, the last urban stream restoration tool involves the judgment of a fishery biologist to determine if downstream fish barriers exist, whether they can be removed, or whether selective stocking of native fish is needed to recolonize the stream reach.

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Chapter 2

ECOLOGICAL ASSESSMENT OF MEDITERRANEAN STREAMS AND THE SPECIAL CASE OF TEMPORARY STREAMS

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ABSTRACT

Freshwater ecosystems play a primary role in the biosphere, which justifies interest in the assessment their health or condition. One important component of the biological assessment of stream conditions is evaluating the direct or indirect effects of human activities or disturbances. Most biological assessments are based on the concept of comparing current conditions to natural conditions in the absence of human disturbance.

In Europe, increasing water quantity and quality problems have led to the development of an integrated approach for water management systems, including all water-related impacts, which resulted in the Water Framework Directive (WFD) in 2000. The Reference Condition Approach concept has been adopted by the WFD as it requires the evaluation of the ecological status and this may be expressed as a deviation from the near natural reference condition. According to the WFD, reference conditions should be linked to stream typology, and reference sites should present the full range of conditions expected to occur naturally within a given stream type.

In the Mediterranean streams context, a methodology to select reference sites based on the application of twenty a priori criteria that reflect the characteristics of Mediterranean streams and their most frequent disturbances, has been a critical step for the establishment of Mediterranean reference conditions. Subsequently, the development of a typology of Mediterranean streams resulted in five ecotypes (four permanents and one temporary), showed that stream classification schemes based on environmental variables need to be validated by biological variables.

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The WFD establishes that ecological status must be classified into five quality classes (high, good, moderate, poor and bad) by means of biological indicators. In this context, a study that analysed the sensitivity of diverse macroinvertebrate metrics to a multiple stressor gradient, which includes the pressures in a Mediterranean area, revealed that the IBMWP and ICM-11indices ensure a proper classification of the ecological status of Mediterranean streams. However, much less is known about the physico-chemical quality of these streams.

An important conclusion drawn from studies conducted in Mediterranean streams indicates the difficulty in assessing temporary streams as regards both biological and physicochemical quality. The diverse macroinvertebrate communities within this type of streams, mainly due to the wide variability in hydrology conditions, could justify further division into subtypes. In relation to physico-chemical quality, temporal streams show an important spatial and temporal variability in solute concentrations. Both low surface flow and spatial intermittency reinforce the effects of differences in advection, water residence time, biological community structure, sediment-water interactions or environment redox conditions through stream channels. By way of example, all these factors influence the rates of the biogeochemical processes involved in the nutrients cycle. Consequently, biogeochemical process rates and reactions will have a patchy distribution in temporary streams. At the same time, this local stream-channel environmental conditions change over time as flow discharge changes, which increases the variability of physicochemical quality on the temporal scale.

1. INTRODUCTION

Freshwater ecosystems play a primary role not only in the biosphere as they support unique and complex ecological communities, but also in the defining and functioning of surrounding terrestrial ecosystems. They also play a key role in fresh water as a human resource. Consequently, there has been growing interest in the design methodologies to assess or monitor the “health”, “status” or “conditions” of freshwater ecosystems (Bailey et al., 2004).

In the 20th century, environmental problems have increased from sewage discharge in the first decennia to today’s climate change (Verdonschot, 2000). Traditionally, water quality programs were mainly based on chemical and physico-chemical parameters, but recently there has been a trend towards the adoption of biological methods (bioassessment) to assess river conditions because effects on biota are usually the final point of environmental degradation and pollution of rivers (Norris and Thoms, 1999).

Biological assessment is defined as the systematic use of biological responses to assess changes in the environment for the purpose of using this information to define a water quality control program (Matthews et al., 1982). The biological response is measured by using biological indicators (Verdonschot and Moog, 2006). Benthic macroinvertebrates have been widely used as indicators of water quality in river management because, apart from their many advantages over using other organisms, they are affected not only by natural changes in the rivers, but also by the chemicals and physical factors induced by human activities (i.e., Rosenberg and Resh, 1993).

The bioassessment of freshwater ecosystems has evolved through a variety of approaches (Bailey et al., 2004). In the first decade of the 20th century, biological assessment mostly used simple straightforward techniques relating to organic waste pollution. Of these methodologies, the saprobic approach and indices have been widely used in many central and eastern European countries (Verdonschot and Moog, 2006).

The term river health has been widely used in ecological literature as a general approach to freshwater ecosystems conditions. Assessment of river health involves comparisons. Indicators thought to represent river health are generally compared between sites that are thought to be similar in the absence of degradation (Norris and Hawkins, 2000). In this context, a new biological assessment approach emerged; the Reference Condition Approach (RCA; Bailey et al., 2004). This approach measures the variability in the biota among sites in Reference Condition (RC). The RCA has three key features: a) it defines and quantifies ecosystem health, b) it explains some variations among healthy ecosystems, and c) the deviation of a test site from RC is a measure of the effect of stressors on the ecosystem. There are different interpretations of the term "Reference Condition" (see Stoddard et al., 2006), but one of the multiple definitions is "the condition that is representative of a group of minimally disturbed sites (reference sites) organized by selected physical, chemical, and biological characteristics" (Reynoldson et al., 1997). With the RCA, an array of reference sites characterises the biological condition of a region. Then a test site is compared to an appropriate subset of reference sites.

Two different widely used procedures for the establishment of RC are multimetric and multivariate methods. These approaches are similar in terms of their data collection methods, but differ in the way reference sites are selected, test sites are classified and test site assessments are made (Resh et al., 2000). On the one hand, multimetrics are attractive because they produce a single score that is comparable to a target value and they include ecological information. On the other hand, multivariate methods offer the advantage of requiring no prior assumptions in either creating groups out of reference sites or comparing test sites with reference groups (Reynoldson et al., 1997).

2. Ecological Assessment of European Streams: The Water Framework Directive

In Europe, increasing water quantity and quality problems have led to the development of an integrated approach for water management systems that includes all water-related impacts. This resulted in the amendment of the Water Framework Directive in 2000 (WFD2000/60/EC; European Commission, 2000) (Achleitner et al., 2005).

One of the main objectives of this Directive is to achieve "good ecological status" in all water bodies of the European Union until 2015 by applying an integrated approach for water management and avoiding the deterioration of the present status of water bodies.

The RCA has been adopted by the WFD as it requires the evaluation of Ecological Status evaluation and this may be expressed as a deviation from the near-natural reference conditions. According to the WFD, RC should be linked to stream typology, while reference sites should present the full range of conditions expected to occur naturally within a given stream type. Ecological status is shown as a new tool to measure the ecosystem functioning and health integrity of aquatic systems, and acts as a useful tool for its management and improvement (Munné and Prat, 2009).

The ecological status measure in this directive acknowledges the importance of aquatic biota as well as the hydromorphological and physico-chemical quality elements for its assessment. Before the WFD, water quality monitoring programmes in most EU Member States were based mainly on chemical and physical variables. Only around half the programmes included biological parameters in their river quality assessments and classifications (Hering et al., 2003).

In recent years, several research projects and working groups have published useful studies into the assessment of streams in Europe and the proper implementation of the WFD. For example, the AQEM (Hering et al., 2004) and STAR projects (Buffagni and Furse, 2006) concentrated on the use of macroinvertebrates as a biological quality element, while the REBECCA project (Solheim and Gulati, 2008) focussed on the relationships between chemical status and ecological status of surface waters. On the other hand, The ECOSTAT working group (CIS WG 2.A. ECOSTAT, 2004) established very useful guidance to support the implementation of the WFD.

2.1. Definition of Stream Typology

Classification of freshwater ecosystems is essential in the development of biological assessment frameworks (Gibson et al., 1996; Gerritsen et al., 2000). Ideally, classifications should identify groups of undisturbed (reference) sites where comparable biological communities might be found, thereby partitioning the natural variation of measurements of ecological quality (e.g., Resh et al., 1995; Dodkins et al., 2005).

In Europe, there has been renewed interest in the regionalization or typology of aquatic ecosystems with the publication of the WFD. Many classification schemes have been used to test the concordance between landscape patterns and the structural and functional aspects of biological communities. For instance, two distinct methods have been used to classify sites: a bottom-up approach using biological communities and a top-down approach based on environmental criteria. The WFD proposes the latter approach, and acknowledges two systems for river classification (Systems A and B). Both systems are based on the division of Europe into ecoregions proposed by Illies (Illies and Botosaneanu, 1963). System A differentiates waterbodies according to the classes determined by using three environmental descriptors: altitude, size and geology, whereas System B is based on five obligatory (latitude, longitude, altitude, size, geology) and several optional variables.

Verdonschot and Nijboer (2004) established three major stream types in Europe: Mountain, Lowland and Mediterranean. However, this classification may be insufficient for capturing the variability present in streams influenced by the Mediterranean climate (Sánchez-Montoya et al., 2007). Furthermore and in the ECOSTAT working group context, different Geographical Intercalibration Groups (GIG), which have been developed in five areas of Europe with similar natural conditions (Nordic, Central/Baltic, Alpine, Eastern Continental and Mediterranean), have established different river types in each area.

2.2. Establishing Reference Conditions

The European WFD establishes the need to define stream type-specific reference conditions to identify high ecological status. Many authors have defined the term RC (e.g. Davies, 1994; Hughes, 1995; Reynoldson et al., 1997; Chovarec et al., 2000; Bailey et al., 2004; Stoddard et al., 2006), and they all emphasise that this condition corresponds to a state of very low environmental pressure or degradation. Although several methods have been proposed to establish the RC (extensive spatial survey, predictive modelling, historical data, paleo-reconstruction and expert judgement), extensive spatial surveys are one of the most widely used approaches in rivers and streams (e.g., Wallin et al., 2003). Following this method implies that a sufficient number of undisturbed or minimally disturbed sites are available in a specific stream type to establish the RC by statistic measures, such as median values or arithmetic means. Irrespectively of the method followed by the EU Member States, the WFD expects the approaches to provide a high level of reliability.

The recognition and selection of reference sites is a critical step in the design of the extensive spatial surveys to establish RC (Reynoldson and Wright, 2000; Bailey et al., 2004) because these sites are the baseline for comparing test sites (e.g., Reynoldson et al., 1997, Norris and Thoms, 1999; Bailey et al., 2004). Reference sites selection is commonly founded on a priori established criteria (Barbour et al., 1996; Reynoldson et al., 1997; Stoddard et al., 2006) based on the different pressures derived from human activities that may affect the ecological status, and can distinguish a reference site from a pressure-exposed site (Hering et al., 2003). Reference sites must fulfil specific operational criteria that easily indicate the absence of exposure stressors (Bailey et al., 2004). The factors chosen to be included in the a priori criteria attempt to define the least amount of environmental disturbance caused by human activities. Thus when establishing criteria, the goal is to explicitly define the reference or acceptable healthy ecosystem (Bailey et al., 2004).

After the a priori selection, site validation must be applied to confirm and refine the selection of reference sites (Barbour et al., 1996) because certain kinds of disturbances are difficult to detect with the common screening methods used (Nijboer et al., 2004). A review of the criteria used to select reference sites in different regions carried out by Sánchez-Montoya et al. (2009a) shows that the most relevant criteria for streams and rivers relate to riparian vegetation, diffuse and point sources, channel morphology, and hydrological conditions and regulations. However other aspects such as introduced species, biomanipulation, presence of wildlife, catchment vegetation, and macroinvertebrate and habitat composition were not considered by most of the reviewed authors.

After the selection of reference sites, the RC of a specific metrics or parameters could correspond to the median value (reference value) of all the reference sites belonging to a specific type (Pollar and van de Bund, 2005).

2.3. Quality Class Boundaries and Macroinvertebrare Metrics

According to the WFD, ecological status must be classified into five quality classes (high, good, moderate, poor and bad). In order to ensure comparability of such monitoring systems, each Member State shall express their result as ecological quality ratios (EQR) to be able to classify ecological status. These ratios shall represent the relationship between the values of the observed biological parameters and the values of these parameters in the reference conditions (reference value). The ratio shall be expressed as a numerical value of between zero and one, where high ecological status will take values close to one, otherwise values close to zero will represent bad ecological status (European Commission, 2000; 2003). The boundary between good and moderate status is especially important because it sets the targets for the restoration plans within the programmes of measures of those water bodies that fail the environmental objectives of achieving good ecological status (Heiskanen et al., 2004).

The definition of class boundaries is a crucial step for implementing the WFD (e.g., Buffagni and Furse, 2006). The 25th percentile of the distribution of the reference values used to be selected as the upper anchor that defines the class boundary between high and good. The best adjustment of the different regressions (e.g., linear or exponential) by means of the adjusted regression coefficient (r^2) between the metric and a specific or multiple stressor gradient must be studied. When the best adjustment between the metric and the stressor gradient is achieved by linear regression, the width of the four remaining classes is evenly spaced over the remaining interval (European Commission, 2003). If, however, the relationship is not linear, an equitable division of the EQR is not appropriate. In such cases, the stressor gradient value corresponding to the class boundary between high and good (established by the 25th percentile) is calculated. Starting from this value, the rest of stressor gradient is divided into four equitable fragments, and the class boundary is the EQR value which coincides with these four established intervals.

The European Commission suggests the use of multimetric approaches for establishing ecological status (Hering et al., 2006), and proposes the use of multimetric indices to detect diverse types of pressures (i.e., organic, hydromorphological alteration, acidification, etc.). The multimetric approach attempts to provide an integrated analysis of the biological community of a site by deriving a variety of biological measures and knowledge of a site’s fauna (Karr, 1981; 1999).

In addition, predictive models (multivariate approach), based on the simulation of the biological community using discriminating environmental variables, have also been developed in Europe; for example, SWEPAC (Johnson and Sandin, 2001), PERLA (Kokeš et al., 2006) and MEDPACS in Mediterranean streams in Spain (Poquet et al., 2009).

Finally, the use of functional trait approaches as one of the most promising tools emerging for biomonitoring freshwater ecosystems must be emphasised (Menezes et al., 2010). Along these lines, Pollard and Yan (2010) indicate that the use of trait-based measures have two potential strengths; traits are thought to be universal across large spatial scales, and they provide a foundation for understanding how functional traits characteristics are expected to respond to environmental gradients.

Those authors demonstrated in this recent work that the studied trait-base assemblage metrics, clinger relative richness and abundance were more consistently associated with a pressure gradient, in this case a sediment gradient, than the comparable identity-base metrics, EPT relative richness and abundance.

3. Ecological Assessment of Mediterranean Streams in Spain

In Spain, the transposition of the WFD in December 2003 implied the adoption of ecosystemic management of water ecosystems. Most of Spain has a Mediterranean climate and in Spain, as in other Mediterranean regions, water has been one of the limiting factors of economic activity for many years, even in large rivers (Prat and Munné, 2000).

In general, European Mediterranean streams have a long history of flow regulation, intensive agriculture land use and diverse sources of pollution (e.g., Gasith and Resh, 1999). This intense exploitation of water has meant that the natural flow regimen and its quality are significant disturbed following an increasing gradient from headwater to the mouth in most streams and rivers. In this context, Mediterranean streams in Spain are not an exception (see Sánchez-Montoya et al., 2005).

The Mediterranean climate is characterised by seasonal variation in precipitation and temperature, with hot, dry summers and cold, wet winters. The discharge regime of Mediterranean-type streams follows this rainfall pattern and, consequently, exhibits both seasonal and annual variability (Gasith and Resh, 1999). Therefore, streams in Mediterranean areas are shaped by sequential, predictable, seasonal events of flooding and drying over an annual cycle. These natural high disturbance levels result in the ubiquitous presence of resilient and relatively persistent macroinvertebrate communities (Aguiar et al., 2002).

In this context, the Guadalmed Project (1998-2005) focusses on evaluating the ecological status of Mediterranean rivers in accordance with the WFD (Prat, 2004), and has shed light on the assessment of Mediterranean streams. Although the multimetric approach has proved an important part of the project, a predictive model was also developed, namely MEDPACs (Poquet et al., 2009).

It is likely because of the novelty in biological assessments in European Mediterranean countries that most of the studies done in this context refer to biological quality elements, and specifically to benthic macroinvertebrate communities, while hydromorphological and physico-chemical elements received less attention. In line with this, this chapter concentrates on biological assessment by using macroinvertebrates as a biological quality element.

3.1. Mediterranean Stream Types and Their Concordance with Macroinvertebrate Assemblages

In Spain, there has been a tradition of regionalization studies (see Margalef, 1983; García de Jalón and González del Tánago, 1986). Recently, a typology for all Spanish territory has been developed by CEDEX (2005), a Spanish civil engineering agency. This typology, which uses system A as suggested by the WFD, classified Spanish watercourses into 33 different stream types. Exclusively in the Mediterranean catchments context, there are also several works. Vidal-Abarca et al. (1990) and Munné and Prat (2004) developed regionalization processes in two different Mediterranean catchments. The latter work tested the two WFD typology approaches (Systems A and B), and concluded that the ecotypes obtained by System B better reflect ecological processes because key environmental factors, such as hydrological and climatic variables, are considered.

In the Guadalmed project context, Bonada et al. (2004) did a preliminary work to develop an optimal typology for Spanish Mediterranean streams. Based on this work, a second regionalization was developed (Sánchez-Montoya et al., 2007) which includes a larger number of sites and new environmental variables.

All these aforementioned works used the top-down approach based on environmental criteria for river classification. However, it must be emphasised that only the typology by Sánchez-Montoya et al. (2007) studied the macroinvertebrate reference communities in each environmental stream type for the purpose of analysing the concordance between environmental and biological classifications. This study was carried out in 33 catchments located along a latitudinal, thermal and pluviometric gradient on the Spanish eastern coast and the Balearic Islands (Figure 1). The study area is influenced by the Mediterranean climate, with significant spring and autumn rainfalls. A total of 162 sites were used to define stream ecotypes (environmental types). The selected sites were considered the least disturbed reaches in the chosen streams, ranging from totally undisturbed to moderately disturbed.

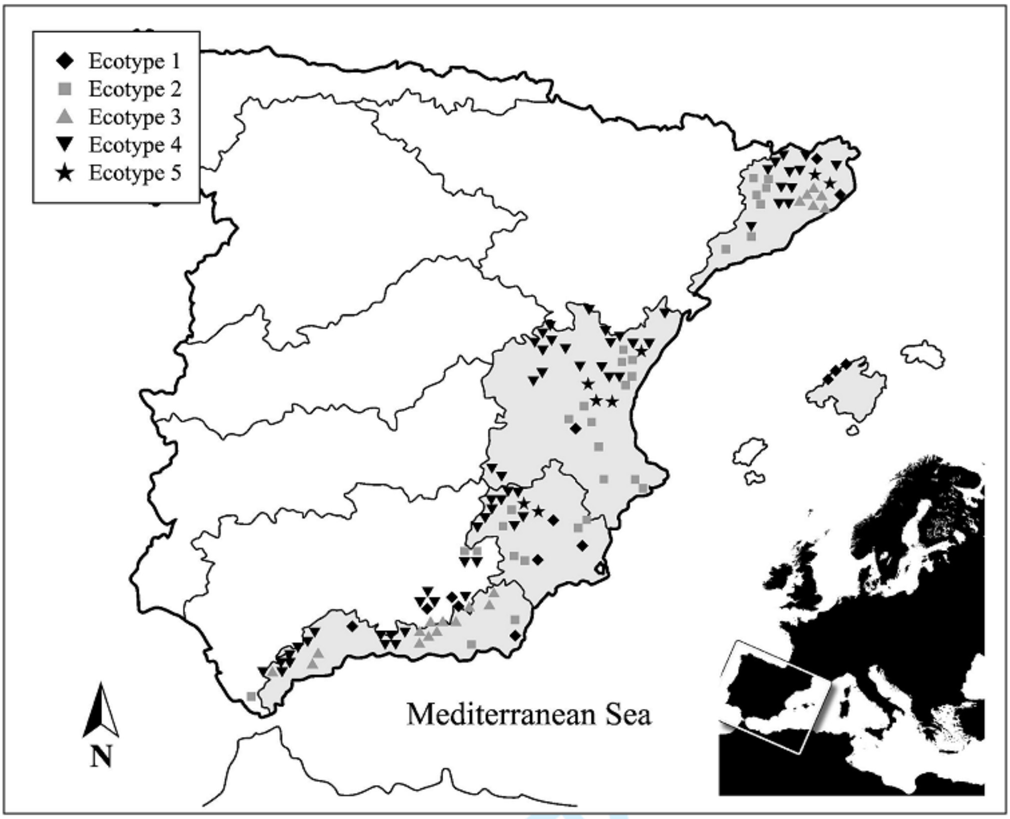

Figure 1. Spatial distribution of the five environmental ecotypes (n=162) according to system B. (from Sánchez-Montoya et al., 2007).

To define stream ecotypes, three of the five obligatory variables proposed by the WFD for System B (altitude, geology and size) were considered. Moreover, 13 optional variables were analysed that could provide useful information about Mediterranean streams, which were grouped into four classes: hydrological, morphological, geological and climatic (Table 1). All these environmental variables used were not influenced by human activities.

After removing two variables because they highly correlated with other variables, a principal components analysis (PCA) was performed on the environmental variables groups. Two new variables (Axes 1 and 2) were obtained for each group of environmental variables. Later, a K-means analysis was performed using the derived PCA variables to classify sites into stream ecotypes. This analysis concluded that the five-group classification showed the greatest ecological and spatial coherence, while a step-wise discriminant analysis (Wilks's Lambda method) revealed the importance of hydrological variables in explaining among-group variance, followed by the geological, morphological and climatic variables. There were five ecotypes: four were perennial and one was temporary. Table 2 shows the main variables defining the five ecotypes.

Table 1. Environmental variables used to characterise ecotypes. Variables marked with an asterisk were later removed because they were highly correlated with other variables within the same group (from Sánchez-Montoya et al., 2007)

Groups of variables	Factors	Description
Hydrological	Discharge	Average of annual discharge ($m^3\ s^{-1}$)
	Hydrological state	Code for hydrological state (see Table 3)
	Dry period percentage	% of year discharge is equal to 0
Climatic	Temperature	Annual average of air temperature (ºC)
	Precipitation	Annual rainfall (mm)
Geological	Surface carbonate geology	% of coverage in the drainage area
	Surface siliceous geology	% of coverage in the drainage area
	Surface evaporate geology	% of coverage in the drainage area
Morphological	Altitude	Meters above sea level
	Drainage slope	Slope/total drained area
	Distance from the origin*	Meters
	Surface drainage area	Total drainage area (km^2)
	Stream order*	Strahler method

In order to validate these previously defined ecotypes, macroinvertebrate communities from true reference sites were used. Reference sites were selected by applying the Mediterranean Reference Criteria (see Section 3.2 of this chapter). All the sampling sites included in Ecotype 5 (large watercourses) evidenced significant human disturbance, resulting in no reference macroinvertebrate communities being found. As a result, top-down versus bottom-up comparison could not be made for these watercourses.

Macroinvertebrate family composition differed between the four defined Mediterranean stream ecotypes according to the IndVal analysis, where siliceous headwater high altitude streams (Ecotype 3) indicator taxa consisted mostly of EPT taxa commonly found in soft-water high-altitude streams, and Ecotype 2 taxa characteristic of the middle courses of streams with a lentic type habitat. This result reveals that these ecotypes are of some ecological significance.

Table 2. Average and standard deviation (in parentheses) values of some environmental variables used to establish the five Mediterranean stream ecotypes (modified from Sánchez-Montoya et al., 2007)

Ecotypes	% Siliceous	% Carbonate	% Evaporite	Hydrological state	Altitude (m)	Stream order	Definition of ecotypes
1	27 (38)	69 (37)	4 (6)	Intermittent/Ephemeral	645 (523)	1.1 (± 0.3)	Temporary
2	13 (22)	67 (21)	19 (11)	Perennial	446 (266)	1.9 (± 0.8)	Evaporite calcareous at medium altitude
3	97 (7)	3 (7)	0.1 (0.4)	Perennial	1177 (589)	1.3 (± 0.6)	Siliceous headwaters at high altitude
4	9 (17)	88 (17)	2 (4)	Perennial	840 (343)	1.7 (± 0.9)	Calcareous headwaters at medium and high altitude
5	20 (31)	70 (28)	9 (4)	Perennial	239 (187)	4.3 (± 0.7)	Large watercourses

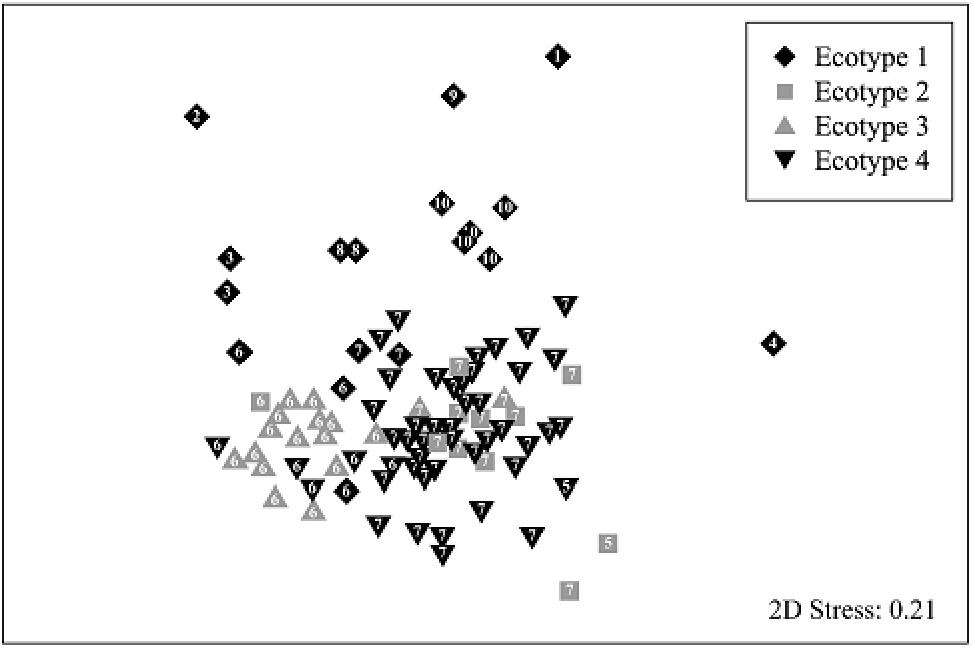

Figure 2. Non-Metric Multidimensional Scaling (NMDS) ordination of the reference sites based on the Bray-Curtis similarities. Labels identify the stream groups defined by UPGMA, and symbols identify ecotypes obtained by System B (from Sánchez-Montoya et al., 2007).

The top-down classification, performed by the UPGMA classification using Bray–Curtis distances, discriminated 10 biological groups. The majority of biological groups were small and belonged to temporary streams, and the remaining ecotypes presented a good correspondence with the biological group. Therefore, the main difference between the communities defined from the top-down versus the bottom-up approaches was found for Ecotype 1, or temporary streams, as shown by the non-metric multidimensional scaling (NMDS) ordination of the reference sites (Figure 2). Finally, the analysis of similarities (ANOSIM) revealed significant differences in the assemblage composition among the four stream ecotypes. Therefore, it can be concluded that these Mediterranean ecotypes, along with the presented ecological coherence, are suitable for performing an intercalibration of the quality class boundaries in the WFD context.

3.2. Selection of Reference Sites in Mediterranean Streams

In the Guadalmed project context, a list of 20 a priori criteria was proposed (Table 3) to ensure the proper selection of reference sites in Mediterranean streams based on their characteristics and the most frequent disturbances (Sánchez-Montoya et al., 2009a). These criteria, defined in this work as Mediterranean reference criteria (MRC), belong to six different categories: riparian vegetation zone, introduced species, point sources of pollution, diffuse sources of pollution, river morphology and habitat conditions, and hydrological conditions and regulation. The 20 apriori criteria include a wide range of human uses and disturbances on streams in our study area, as well as some general aspects on naturalness that must be present if a site is to be considered a reference site. The spatial scale of application was sub-basin (upstream to the site) for all the criteria. With this approach, only those sites that fulfil all 20 criteria can be considered reference sites. In order to apply both this screening method and a validation process, the same study area and sites as in the aforementioned regionalization work were studied (162 sites in 33 basins).

After applying the MRC, a good number of the sites (92 of 162, 57%) fulfilled all 20 criteria and were considered reference sites in this a priori selection. Reference sites were found for all stream types, except for large watercourses. In general, the most limiting criteria were those relating to natural land use and to dry land farming, which affected more than 20% of the sites. This finding was expected because, traditionally, agriculture has been a very important productive sector in Mediterranean Europe (Allen, 2001), where dry and irrigated farm lands are significantly more frequent than in other European countries (MIMAN, 2000). Moreover, disturbances caused by the presence of reservoirs, retention of sediments, exotic species and irrigated farming affected a considerable percentage of sites, unlike the canalisation, adequate substrates and grazing criteria which all the sites fulfilled.

In order to refine and confirm the a priori selection of reference sites by applying the MRC, a macroinvertebrate community was used for validation purposes by applying the reliable biological threshold as defined by Alba-Tercedor and Sánchez-Ortega (1988). The threshold was the value of the IBMWP >100, considered a high quality class indicator. For this purpose, the macroinvertebrate community from the reference site was sampled on three sampling occasions (spring, summer and autumn, 2003). Finally, in order to be considered a reference site in the final selection, the a priori reference sites had to fulfil the biological validation criterion on the three sampling occasions.

Table 3. List of 20 criteria proposed for the selection of reference site of Mediterranean streams (from Sánchez-Montoya et al. 2009a)

Elements	Criteria
Riparian vegetation zone	1. Cover and composition appropriate for the type and geographical location of the river. 2. Lateral connectivity between river and riparian corridor is maintained (no cultivation and significant impervious area in riparian zone).
Introduced species	3. No significant impairment by exotic plant or animal species on autochthonous species
Point sources of pollution	4. No dumping of urban effluents 5. No dumping of industrial effluents 6. No irrigation return channel for floodwater farming
Diffuse sources of pollution and land uses	7. Dry land farming < 20% of drainage area (cereal, vineyard and tree crops as olive) and not connected to riparian vegetation zone 8. Intensive irrigated farming < 3% in drainage area (rice field, irrigated vineyard and others irrigated fruit tress) and not connected to riparian vegetation zone 9. Urban use < 0.7 % in drained area 10. Burned vegetation < 7% in drainage area at least seven years ago and not connected to riparian vegetation zone 11. No evidence of intensive use of grazing 12. Natural land uses > 80% in drainage area
River morphology and habitat conditions	13. Representative diversity of substrate materials appropriate for the type 14. No canalization (stream bottoms and stream margins must not be fixed) 15. No transversal structures "dams" (no retention of sediments) 16. No sand or gravel extraction
Hydrological conditions and regulation	17. No water diversion for irrigation or other purpose 18. No alterations of the natural hydrograph and discharge regime (reservoirs, hydroelectric…) 19. No effect of inter-basin water transfer 20. Natural level of groundwater (aquifer not affected by over-exploitation)

As a result of applying the validation process, all 92 a priori reference sites exceeded the threshold value on the three sampling occasions, and were selected as true reference sites in the final selection. This result validates the application of MRC for selecting true reference sites in Mediterranean streams.

3.3. Biological Assessment of Mediterranean Streams: Best Macroinvertebrate Metrics and Quality Class Boundaries

One main WFD approach is the use of biotic indicators (phytobenthos, macrophytes, macroinvertebrates and fish) in stream assessment. However, biological quality monitoring of rivers is not a long-standing tradition in some Mediterranean countries like Spain.

There is a general consensus about the value of macroinvertebrates as water quality biological indicators (Chessman and McEvoy, 1998); therefore, the community structure of benthic invertebrates has been frequently used in environmental monitoring and assessments of aquatic systems (Reynoldson and Metcalfe-Smith, 1992).

Another important point in assessing ecological status is the identification of metrics that can detect one or several types of human perturbation.

Diverse macroinvertebrate indices have been proposed to assess the water quality in Mediterranean streams and rivers (see Munné and Prat, 2009) based on qualitative and quantitative data. Although abundance measures (quantitative data) are explicitly demanded in the WFD, these are known to have high variances and are rarely used in multimetric approaches (Barbour et al., 1999). Particularly in Mediterranean streams and rivers, it has been proved that abundance metrics discriminate worse different levels of degradation than the presence-abundance metrics given the strong fluctuation of the hydrologic regimens in these streams that produce a high variability in taxa abundance (Pinto et al., 2004).

A recent study carried out in Mediterranean catchments in Spain (Sánchez-Montoya et al., 2010) compared the sensitivity of diverse metrics to a multiple stressor gradient in Spanish Mediterranean streams and its influence on the assessment of ecological status for the purpose of selecting the best macroinvertebrate metrics based on their response to this stressor gradient.

Seven qualitative metrics were studied: two single metrics (the number of families (NFAM) and the number of Ephemeroptera, Plecoptera and Trichoptera families (EPT)), three indices (the Iberian Biological Monitoring Working Party Index (IBMWP), the IBMWP value/number of families (IASPT), the Biological Monitoring Water Quality (T-BMWQ), and two multimetrics indices (the ICM-9 and the ICM-11a).

ICM-9 is a multimetric index proposed by the Mediterranean Geographical Intercalibration Group (Med-GIG) as an appropriate index for Mediterranean streams (European Commission, 2007). ICM-11a, also called IMMi-L, developed from the Spanish experience in the Med-GIG, and has been recently developed specifically for macroinvertebrate communities inhabiting Mediterranean river systems (Munné and Prat, 2009). In Spain, the IBMWP index (Alba-Tercedor and Sánchez-Ortega, 1988; Alba-Tercedor et al., 2004) is the most widespread and extensive index employed in Mediterranean rivers; indeed, the Spanish Environmental Ministry proposed this index to assess ecological status in Spanish Mediterranean streams (MMARM, 2008).

For this purpose, a total of 193 sites belonging to 35 catchments on the Iberian Mediterranean coast were studied. These sampling sites represent different grades of degradation, ranging from undisturbed (reference sites) to highly disturbed according to the number of fulfilled reference criteria as defined by Sánchez-Montoya et al. (2009a). Non-reference sites showed a combination of several pressures, of which the most frequent were those relating to non-natural land uses, diffuse sources of pollution and flow regulation.

To create the stressor gradient, a PCA was performed using 27 different variables belonging to the various categories (physico-chemical, land uses, hydrological variables and quality indices) studied for all the sampling sites.

The analysis of the response of the seven metrics to the stressor gradients by linear and exponential regressions revealed that the exponential regression coefficients for all the studied metrics were higher than the linear coefficients, indicating an exponential relationship between metrics and the environmental alteration. The two studied multimetric indices presented higher regression coefficients than the three indices and the two metrics, which depicts a better response to the stressor gradient in the Mediterranean streams under study.

This finding supports the knowledge that a single metric responds well to specific pressures, but that multimetric indices provide an integrated analysis of the biological community and are capable of detecting multiple stressors.

The results obtained in this work support the knowledge that the IBMWP, widely used in Spain, is suitable for biological quality assessment in Mediterranean streams because of the high regression coefficient obtained. Since the European Commission suggests the use of multimetric indices, and based on the results obtained in this work, the use of the ICM-11 multimetric index for Mediterranean streams is also recommended.

The WFD emphasises that the natural variability of quality elements in high status (pristine) needs to be quantified. Therefore, natural variation of macroinvertebrate metrics may be taken into account to select suitable metrics for ecological status assessment. Ignoring natural variability can confound the detection of anthropogenic environmental changes (Clarke and Hering, 2006). Accordingly, a robust biodiversity metrics should show low natural variability if compared to any change taking place as a result of human disturbance (e.g., Johnson, 1998; Sandin, 2003; Springe et al., 2006). As the spatial variability of quality elements is reduced by the type-specific reference condition approach, temporal variability is the main aspect to be analysed.

Temporal variation in macroinvertebrate communities is often high in stream systems given flow and habitat variation (e.g., Feminella, 1996; Townsend et al., 1997), among other factors. Such fluctuations are expected to be higher in areas with relatively high environmental variability, which is the case of those areas influenced by a Mediterranean climate (Bêche and Resh, 2007).

This subject was studied in the work of Sánchez-Montoya et al. (2009b), where both the seasonal and interannual variations of the macroinvertebrates communities in reference sites and their effect on macroinvertebrate metrics in Mediterranean streams types were studied. To detect seasonal changes, 88 reference sites belonging to four Mediterranean ecotypes defined by Sánchez-Montoya et al. (2007) in 23 catchments were used, and a subset of 14 reference sites belonging to two Mediterranean ecotypes were studied for interannual change. Eighteen macroinvertebrate metrics belonging to different categories (richness, index, multimetric index, tolerance/intolerance and diversity), which are widely used in assessing running waters, were calculated for all the reference sites on each sampling occasion.

In order to select robust metrics, the calculation of the coefficient of variation (CV) revealed that all the metrics for all stream types were low (< 50%). The lower seasonal CV values (< 15.3%) were observed for taxa richness (S) and macroinvertebarate indices (ICM-11, IBMWP and IASPT) in all the stream types. In general, the seasonal CV values for each metrics were similar in all four stream types. Therefore, these results corroborate the suitability of both the IBMWP and ICM-11a given their low natural variability.

It must be taken into account that given the possible differences in the taxonomic composition among the different stream types, class boundaries must be calibrated separately for each stream type. In the study of Sánchez-Montoya et al. (2010), the quality class boundaries for the IBMWP, and for the ICM-11a in different Mediterranean stream types, were calculated.

4. Ecological Assessment of Mediterranean Temporary Streams

Temporary rivers are not restricted to arid regions and occur in most terrestrial biomes between 84ºN and S latitude (Larned et al., 2010). Notwithstanding, temporary and ephemeral streams are the dominant drainage systems in arid and semiarid zones (Boulton and Suter, 1986).

Temporary streams may be defined as bodies of fresh water that undergo a recurrent dry phase of varying length that is sometimes predictable in terms of both its time of onset and duration (Williams, 1996). Therefore, temporary rivers are characterised by repeated onset and flow cessation of flow, and by complex hydrological dynamics in the longitudinal dimension which influence biotic communities and nutrient and organic matter processing (Larned et al., 2010).

It must be considered that temporary streams are subdivided into intermittent and ephemeral streams depending on their dry-period condition. According to the classification of Uys and O'Keeffe (1997), flow disappears in intermittent streams, but still water continues to be present; yet conversely to ephemeral streams, water disappears and dry stream beds occur.

In temporary streams, the main factors influencing the aquatic assemblages include the frequency and duration of disturbance events and their predictability (e.g., Stanley et al., 1997; Gasith and Resh, 1999).

Unlike temperate rivers, a feature of Mediterranean and other temporary streams is hydrological intermittency (Gasith and Resh, 1999; Acuña et al., 2005), which strongly influences the structure and functioning of aquatic ecosystems. As in other arid streams (Fisher et al., 1982) under base flow conditions, temporary Mediterranean streams usually have a discontinuous surface flow through a series of relatively permanent segments separated by dry stretches. For example, in the province of Murcia in southeast of Spain, which is the driest part of the Iberian Peninsula and probably of Europe, approximately 98% of mapped watercourses are temporary and ephemeral streams (Gómez et al., 2005).

This intermittent surface flow pattern is obliterated during high-flow periods (usually after rainstorms) when the surface flow along the reach channel is continuous. The length of dry stretches normally increases in summer when evapotranspiration increases and discharge declines.

Although these systems are the most abundant fluvial ecosystem types in many Mediterranean areas, they are less understood than permanent streams given the complexity of these systems, and since this hydrological complexity makes the biological, hydromorphological and physico-chemical evaluations of the ecological assessment of temporary streams difficult.

4.1. Biological Assessment of Mediterranean Temporary Streams

In the biological assessment context, the study of the effect of hydrology on macroinvertebrate metrics in temporary Mediterranean streams is quite a recent task (e.g., Álvarez et al., 2001; Morais et al., 2004; Bonada et al., 2007, Sánchez-Montoya et al., 2007; Argyroudi et al., 2008) given hydrological complexity and methodological difficulties.

However, the WFD urges the ecological assessment of temporary streams apart from permanent streams.

Although the responses of macroinvertebrate communities to flow permanence have been widely reported, no consensus has been reached as to there being differences between macroinvertebrate compositions in temporary and permanent streams. Some authors detected no significant differences in the biological compositions of these systems (e.g., Boulton and Suter, 1986; Williams, 1996; del Rosario and Resh, 2000), while others found great differences in their macroinvertebrate communities in Mediterranean regions (e.g., Maamri et al., 2005; Sánchez-Montoya et al. 2007). In this frame, and also in the WFD context, the first problem arises when a typology of Mediterranean streams that considers temporary streams must be established.

The study of Sánchez-Montoya et al. (2007), carried out in reference Mediterranean streams in Spain, showed different macroinvertebrate family compositions (using abundance and presence/absence data) between temporary and remaining permanent streams belonging to three predetermined ecotypes (see Figure 2), which justifies the division of temporary streams as a specific ecotype. This distinction was also made by the ECOSTAT intercalibration group for Mediterranean rivers (Buffagni and Furse, 2006; European Commission, 2005) by establishing a typification for the Mediterranean at the European level where five river types were determined (four permanents and one temporary; see Munné and Prat, 2009). Nonetheless to our knowledge, this classification does not include a validation using biological communities.

Generally, on the other hand, permanent streams present higher values of taxon richness and some macroinvertebrate metrics than neighbouring temporary streams (Williams 1996, Meyer and Meyer 2000, Morais et al., 2004). Buffagni et al. (2009) reported that an increase in lentic conditions is associated with a decrease in the quality metrics value. This finding must be considered when the reference values of temporary streams are established because this “natural” low value may be confused with anthropogenic perturbation and lead to underestimating ecological quality.

In this frame, a study into macroinvertebrate communities in reference conditions in different Mediterranean stream types (Sánchez-Montoya et. al., 2009b) confirmed the aforementioned finding since the median and 25th percentile of the studied metrics values (Number of Ephemroptera, Plecoptera and Trichopetra, the IBMWP index, the IASPT index and Number of families) were lower for temporary streams than for the three remaining types of permanent streams (Figure 3).

The temporary stream type included in this work underwent intense drought periods, and pool habitats frequently disconnected from riffles predominated. The degree of habitat connectivity may influence assemblage richness and diversity, and low taxa richness should be expected in disconnected habitats (Bonada et al., 2006) because low connectivity results in low richness given the exchanges of matter, energy and organism that are constrained between parches (Ward et al., 1999). Besides, it must be considered that spatial heterogeneity increases from ephemeral to intermittent and to permanent sites because higher flows increase available refuges and substrate diversity (Bonada et al., 2007), and higher richness taxa are expected with increases in this heterogeneity. As disconnected pools could be a refuge for some tolerant lentic macroinvertebrates from desiccation (i.e., OCH) but probably not for rheophilic macroinvertebrates (i.e. EPT) (e.g. Brown and Brussock, 1991), lower taxa richness (family level) and EPT richness (relating to OCH richness) values in temporary streams are expected (Bonada et al., 2006; Sánchez-Montoya et al., 2007). Regarding the IBMWP, the high EPT families score in this index (mean value = 8.4) compared with the OCH families (mean = 4.9), explain the lower IBMWP values in temporary streams in the RC. In conclusion, lower metric values in the RC in temporary streams, if compared to permanent streams, must be attributed to natural differences in macroinvertebrate communities.

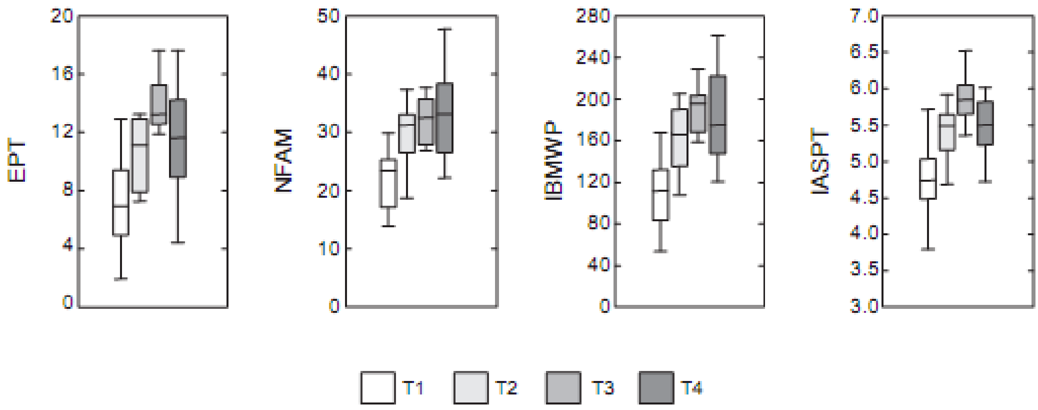

Figure 3. Boxplots for NFAM, EPT, IBMWP and IASPT values of four stream-types (T1: Temporary stream, T2: Evaporite calcareous permanent stream, T3: Siliceous permanent headwaters and T4: Calcareous permanent headwaters). Boxes are interquartiles ranges (25th percentile to 75th percentile), rangers bars show maximum and minimum values and lines represent medians (modified from Sánchez-Montoya et al., 2009a).

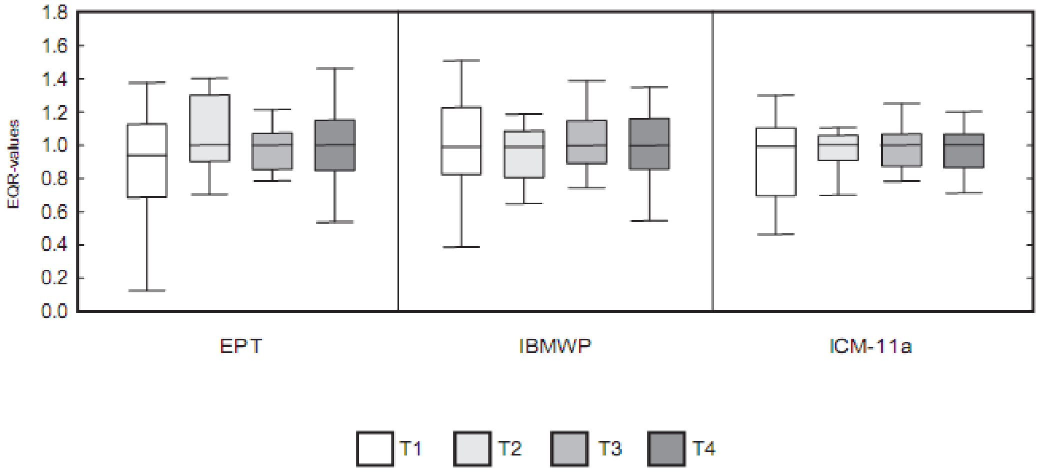

Figure 4. Distribution of the EQR values of reference sites in four stream types (T1: Temporary stream; T2: Evaporite calcareous permanent stream, T3: Siliceous permanent headwaters and T4: Calcareous permanent headwaters). The median values (central line), 25th and 75th percentile values (box) and the maximum and minimum values are shown (modified from Sánchez-Montoya et al., 2010).

Another important finding that arises from the study of Sánchez-Montoya et al. (2007) must be considered to classify quality-class boundaries into different types: while permanent ecotypes generally hold an exclusive macroinvertebrate community, temporary streams include many biological groups. These findings are expected because all the sites belonging to the temporary ecotype were intermittent or ephemeral, and hence underwent intense drought, usually in summer. Therefore, they underwent expansion and contraction processes which could affect the physico-chemical properties of water, and subsequently the composition of macroinvertebrate assemblages (e.g., Resh et al., 1988; Williams, 1996; Lake, 2000; Acuña et al., 2005). An additional point to consider is that drought events can lead to the stream channel drying up partially or completely; in fact, the studied intermittent streams showed different stages of drying and the presence of different macrohabitats (from a riffle-pool sequence to isolated pools) that influence the composition of macroinvertebrate assemblages (e.g., Logan and Brooker, 1983; Bonada et al., 2006).

This wide variability in the reference macroinvertebrate community in temporary stream was confirmed when the EQR (Ecological Quality Ratio) reference values of the different metrics and indices were analysed (Sánchez-Montoya et al., 2010) owing to the higher variability of the EQR reference values in the temporary stream type than in the remaining permanent types. Figure 4 illustrates the distribution of the reference sites EQR values in four stream types for three metrics (EPT, IBMWP and ICM-11a). In all the metrics, the variation of the temporary stream types was greater than the rest of the permanent streams.

Several studies have indicated a possible difference between intermittent and ephemeral streams based on macroinvertebrate composition data and metrics (Bonada et al., 2007; Argyroudi et al., 2009). Furthermore, Meyer et al. (2003) detected decreasing species richness of benthic macroinvertebrate communities at increasing intermittency in karstic stream systems, which was not attributed to physico-chemical or structural variables.

An additional important factor to consider as a source of variation in Mediterranean temporary streams is the high salinity of some streams as flow crosses salt-rich rocks. These saline streams have marked a difference in taxonomic macroinvertebrate composition compared to their freshwater counterparts in the same regions (Moreno et al., 1997; 2001; Mellado et al., 2008, Arribas et al., 2009).

In conclusion, the wide variability in biological communities in temporary streams may justify a further division of this ecotype into subgroups to ensure the proper application of the WFD methodology. This subdivision may relate to the varying permanence of superficial flows in intermittent and ephemeral streams and natural salinity.

4.2. Physico-Chemical Assessment of Mediterranean Temporary Streams

Specific physico-chemical quality elements are required for the determination of high and good status. For the other status classes, the conditions of the physico-chemical elements are expected to be consistent with the achievement of values for the biological quality elements. Estimates of the physico-chemical reference conditions for the developed regions should reflect the natural characteristics of the watersheds of interest. According to the WFD, the physico-chemical parameters supporting the biological elements are: thermal and oxygenation conditions, salinity, nutrient conditions and acidification status.

The use of physico-chemical indicators for ecological assessments entails having to previously establish the physico-chemical reference conditions, as the same as the biological indicators. Chemical parameters (pH, conductivity, dissolved oxygen or nutrient content, among others) are markedly affected by both biotic and abiotic variables which, in turn, may change through space and time. Variation in aquatic nutrient loading relates to watershed geology, vegetation, atmospheric inputs and land use (Hynes, 1975), but variation in nutrient concentration has also been observed on smaller scales in aquatic systems (e.g., Pringle, 1990; Martí and Sabater, 1996; Dahm et al., 1998; Dent and Grimm, 1999; Kemp and Dodds, 2001; Lewis et al., 2007). However, despite the fact that variation at this scale is likely to affect the local abundance and distribution of organisms, microbial activity and primary production (Pringle, 1990; Sabater et al., 2000; Rees et al., 2006) this scale of variation is less well studied. Physico-chemical parameters are subject to spatial and temporal variability in all aquatic ecosystems, rivers and streams. However the grade of variability increases when we analyze temporary streams.

On the other hand, it has been widely acknowledged that establishing physico-chemical criteria in streams is acutely difficult because a considerable amount of monitoring data are required to refine indications about them (Dodds and Welch, 2000). In this sense, the handicap of temporary streams lies in that they are less studied than more temperate streams. Despite their abundance, temporary streams have been historically neglected by ecologists (Larned et al., 2010). This section focusses on considerations into the spatial and temporal variability in solute concentrations, mainly nutrients, as an important element of chemical status.

. In temporary streams, fluctuating low flows create a distinctive mosaic of local environmental conditions (advection, water residence time, oxidising and reducing conditions, sediment-water interactions, etc.) and biological processes (Stanley and Boulton, 1995; Valett et al., 1996; Lewis et al., 2006; Henry and Fisher, 2003; Lillebo et al., 2007) which may directly affect the chemical environment. Organic matter decomposition, nitrification (a source of nitrate), denitrification (a sink of nitrate) or biological uptake, are all biological processes affected by changes in local stream-channel environmental conditions, which are also implicated in the water nutrient content in un-impacted streams (e.g., Sabater et al., 2000; Strauss et al., 2002; Arango et al., 2008).

Besides biotic factors, both low surface flow and spatial intermittency reinforce the effect of abiotic heterogeneity through stream channels that affect solute concentrations. For instance, Gómez et al. (1994) found a direct relationship among the patchy distribution of stream Fe-rich sediments, soluble P concentrations and algae distribution in an arid stream wetland of southeast Spain.

Since flow and advection in temporary streams are lower, and as opportunities for biotic and abiotic interactions are more plentiful, spatial variability in solute concentrations increases, as several studies carried out in arid zones have demonstrated (e.g., Lewis et al., 2006; 2007). In a temporary Mediterranean stream in southeast Spain (the Chicamo stream), Gómez et al. (2009) analysed differences in spatial and seasonal variability in the N and chloride concentrations between reaches with differing hydrological regimes (permanent *vs.* intermittent), under different hydrological conditions (baseflow, high flow and drought). These authors discovered that both, spatial and seasonal variability were higher in the intermittent reach than in the permanent reach as a result of the surface flow's more fluctuating conditions. Then, it is important to consider the acute spatial variability in the physico-chemical properties of temporary streams when monitoring programmes have to be designed to correctly select the number and locations of survey points.

Besides in temporary streams, the local stream-channel environmental conditions that influence spatial variability in chemical conditions may change over time as flow discharge changes (Stanley and Boulton, 1995; Martí et al., 1997; Dahm et al., 1998; Dent and Grimm, 1999; von Schiller et al., 2008a), and increase global chemical variability. In fact, spatial variability has been demonstrated to increase close to summer as flow discharge diminishes. In a 10-km stretch of Sycamore Creek (Arizona, USA), variability in N concentrations increased over successional time as flow discharge decreased and spatial patchiness increased (Dent and Grimm, 1999). Similarly, Gómez et al. (2009) also described an increase in spatial variability of the chloride and nitrate-N concentrations in the Chícamo stream (southeast Spain) during drought.

In addition to the spatial variability problem, solute concentrations tend to be higher in temporary streams as a result of the low surface discharge and the longer water residence time (Gómez et al., 2009). Under these hydrological conditions, and because of limited dilutions and lower surface stream areas, any source of solute inputs (such as bedrock) directly affects stream water quality. In temporary streams, this effect is more patent than in those streams with a higher surface discharge. This situation is, in turn, enhanced close to drought when surface flow decreases (Gómez et al., 2009). Figure 5 presents a conceptual hydrochemistry variability model in Mediterranean arid and semi-arid streams (Vidal-Abarca et al., 2004). This model describes how variability in surface water hydrochemistry is expected to be low under flood conditions given the homogenisation of local environmental conditions as flow discharge increases. In this high surface discharge situation, the chemistry of surface waters, that depend on rainfall volume and intensity, will be a good indicator of watershed geology and land uses. As seasons pass, and as surface flow discharge decreases, hydrochemistry variability increases and reaches its peak close to drought when an intermittent pattern of surface flow appears (spatial fragmentation). At this moment are the to local streambed conditions which determine the hydrochemistry of surface waters. At this moment, hydrochemistry is not a good indicator of watershed conditions.

Lithology is an important source of variation in the physico-chemical properties of surface waters within and across streams. In Mediterranean regions, lithology shows important spatial heterogeneity. In fact in a small area, maybe of several kilometres, different geological materials can be found which determine the heterogeneous bedrock nature of streams. For example, in the province of Murcia, an arid area of southeast Spain, different temporary streams (*ramblas*) have been categorised on a lithological basis (ramblas of marl, limestone and metamorphic basins). These streams differ in structural parameters, hydrology, biological communities and, more importantly, water chemistry terms (Gómez et al., 2005). Differences in water chemistry (mainly sulphate, chloride and nitrate content) as a response to changes in bedrock material can even be observed through the same stream (García, 2005; Gómez et al., 2009). Figure 6 shows the spatial variability in the nitrate-N and chloride concentrations in the Chicamo stream from head to mouth. This temporary stream located in southeast Spain is characterised by two reaches with differing hydrology and lithology (Gómez et al., 2009): a permanent reach in the upper part of the watershed dominated by limestones and dolomites (C2 to C15 in Figure 6) and an intermittent middle-to-low reach in the lower part of the watershed where Miocene-Keuper marls are the dominant lithological material (from C15 to C31 in Figure 6). The effect of both lithology and fluctuating low flows on solute concentrations in the lower part of the stream is evident.

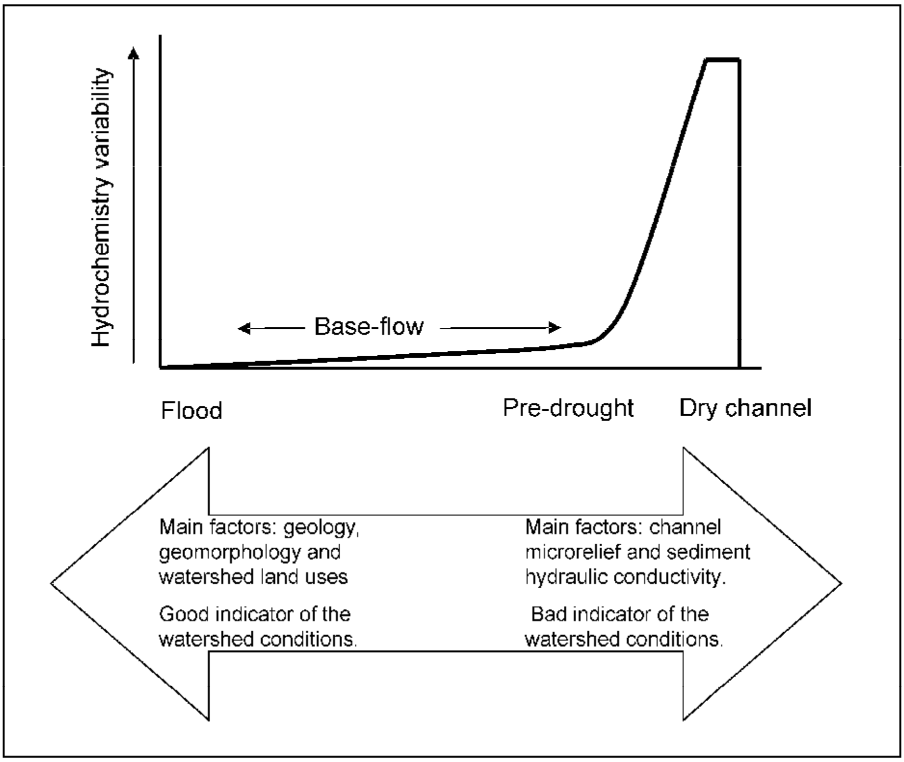

Figure 5. Conceptual model on hydrochemistry variability over seasons in arid and semi-arid streams (modified from Vidal-Abarca et al., 2004).

Other authors also describe the wide range of physico-chemical features that characterise Mediterranean temporary streams, including variability in salt content (mainly NaCl). Temporary streams in the Mediterranean region are characterised by a wide range of conductivity values, with classification ranging from freshwater streams to hypersaline streams (Vidal-Abarca et al., 1995; Moreno et al., 1995; Moreno et al., 2001). Even within saline streams, different categories can be established (e.g., Arribas et al., 2009.) Besides, it has been observed that the temporary streams located in sedimentary watersheds are characterised by a high nitrate-N content. Concentrations of up to 3 mg/l are not unusual in un-impacted reaches or springs (García, 2005; Gómez et al., 2005). Along these lines, other authors have also described high nitrate-N concentrations in Mediterranean streams of sedimentary watersheds (Holloway et al., 1998; Dahlgren, 1994; Holloway and Smith, 2005). These authors have demonstrated that bedrock containing considerable concentrations of fixed nitrogen contributes a surprisingly large amount of nitrate to surface waters in certain California watersheds, and to such an extent that even small areas of these rocks had a profound influence on water quality. Our experimental and field data, obtained in several N-rich arid streams from sedimentary watersheds (*in preparation*), agree with these results.

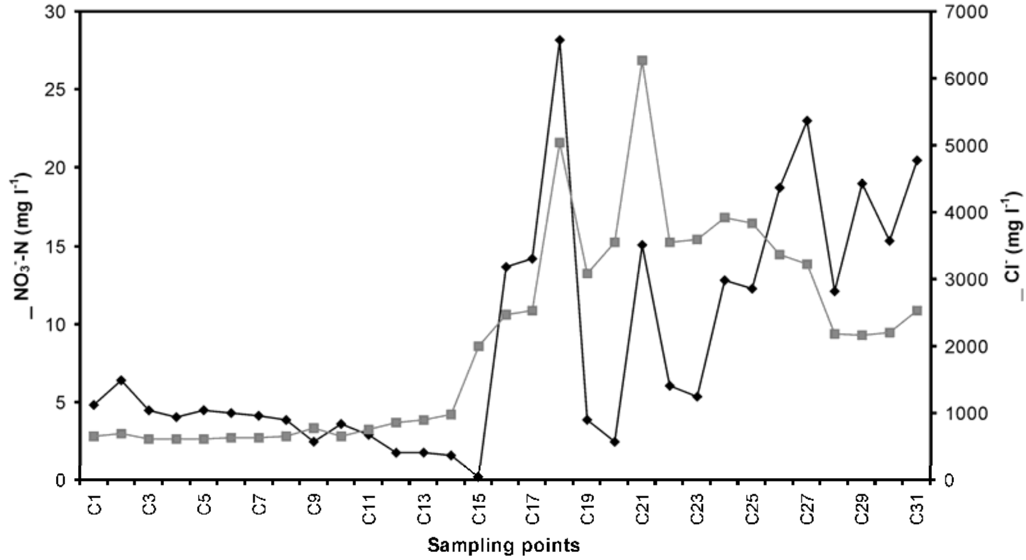

Figure 6. Spatial variation of NO_3^--N (black diamonds) and Cl^- concentrations (grey squares) under base-flow conditions through Chicamo stream (SE of Spain) from head to mouth. C1-C15 are sampling points located in a limestone-dolomite permanent reach, whereas C16 to C31 are located in a marly intermittent reach (from García, 2005).

As regards temperate rivers, diminished surface flows in temporary streams also heighten the influence of vertical (hyporreic) and lateral components (parafluvial and riparian zones) on solute and oxygen concentration. In Sycamore Creek, an arid stream of the Sonoran Desert (USA), increased nitrate concentrations (and phosphorus) in areas where subsurface waters come into contact with the surface (hyporheic upwellings) have been widely documented (Grimm et al., 1991; Valett et al., 1994; Jones et al., 1995; Dent et al., 2001). In this N-limited stream (Grimm and Fisher, 1986), the enhanced nitrate availability produce an increase in primary producers biomass which, in turn, influenced surface water temperature, dissolved oxygen, and dissolved organic carbon (DOC), among other physico-chemical parameters (Fisher et al., 1998). Similarly, Gómez et al. (1995) described peak concentrations of nitrate and ammonium, which were attributable to high nitrogen levels in lateral seepage water and local groundwater discharge, respectively. These authors described changes in N concentrations, which went from 0.31 to 2.02 mg/l NO_3-N and from 0.24 to 1.32 mg/l NH_4-N, along a 20-m reach of a non-impacted arid stream wetland. Because of low discharge, groundwater inputs, such as subsurface flows or lateral seepage, can locally change the local water chemistry by nutrient or salt enrichment or by dilution.

Temporary Mediterranean streams are subject to disturbance by infrequent but intense rainfall, which generates high flow pulses and occasional flash floods, and by drought periods (Vidal-Abarca et al., 1992). These fluctuations disturb sediments and biota that process material (Fisher et al., 1982; Ortega et al., 1991; Stanley et al., 1997; Vidal-Abarca, 2007) which affect solute concentrations. On the other hand, hydrological fluctuations strongly influence nutrient dynamics (Dahm et al., 2003; Bernal et al., 2005; Lillebo et al., 2007). In particular, biotic uptake, an important driver of water nutrient content, may be influenced by changes in discharge (Stream Solute Workshop, 1990; Peterson et al., 2001; Wollheim et al., 2001), as reported in Mediterranean streams (Martí and Sabater, 1996; von Schiller et al., 2008a). Besides, and as a result of lower water velocities and higher water residence times in summer, solute concentrations increase. Gómez et al (2009) described higher N and Cl concentrations in an intermittent reach than in a permanent reach of the same arid Mediterranean stream (the Chicamo stream) under base-flow conditions as a result of differing hydrological conditions. In addition, solute concentrations increased close to summer in the intermittent reach owing to lower surface flow, whereas the permanent reach was hydrologically more stable and solute concentrations did not change (García, 2005). An opposite effect on solute concentrations was described in Rio Calaveras (New Mexico, USA). In this stream, Dahm et al. (2003) described how dissolved organic carbon (DOC) concentrations decreased, as inputs from groundwaters diminished during drought. This change in DOC availability favoured, in turn, autotrophs more than heterotrophs, with the consequent impact on nutrient concentration.

Thus, chemical parameters are expected to be more variable over time in temporary streams where surface flow discharge shows greater fluctuations than in permanent streams.

Table 4. Variability in physico-chemical parameters at different spatial and temporal scales

	Spatial scale			Temporal scale		
	Local variation *	Variation up- to down-reach **	Variation across streams +	Daily variation ++	Seasonal variation ^{x}	Inter-annual variation ^{xx}
Discharge (L/s)	1.2-10.5	0.3-20.0		0.8-3	9.1-70.9-8.4	3.4-11.1-18.1
Conductivity (mS/cm)	11.0-15.1		0.84-10.37	6.4-29.2		12.8-13.6-13.8
Chloride (meq/L)		28.2-138.0	1.64-43.6		56.4-18.6-33.8	
Dissolved oxygen (mg/L)			0-25.4	4.3-20		12.3-8.6-9.1
Oxygen saturation (%)	90-140			40.4-222		129.4-97.4-95.2
Nitrate-N (mg/L)	1.4-3.5	2.3-29.5	0.05-1.8	1.5-3.6	8.8-2.8-5.8	2.8-2.4-2.6
Amonium-N (mg/L)	0.1-0.4	0.01-0.38		0.03-0.1	0.03-0.1-0.01	0.5-0.1-0.09
Orthophosphate- P (µg/L)	4.8-12		15.4-2219.2	0.5-6.0		8.0-9.0-5.0

* Local variation in Chicamo stream under base-flow conditions through a 350 m-reach. Data are the minimum and maximum average values (n= 3) from one date (extracted from Vidal-Abarca et al., 2000).

** Variation in Chicamo stream through a non-impacted 10 km-reach, under base flow condition (extracted from García, 2005).

+ Variation across non-impacted streams located in the province of Murcia under base-flow conditions. Data are the minimum and maximum average values (n=18) from one date (extracted from Moreno et al., 1995).

++ Daily variation in Chicamo stream. Data are average values from 4 dates and 6 points (extracted from Vidal-Abarca et al., 2002). Discharge values of 0.0 (from pools) are discarded.

^{x} Variation in Chicamo stream through seasons: under base-flow conditions, after rainfall and under post-drought conditions. Values are average data from n= 30, 14 and 15 points, respectively (extracted from García, 2005).

^{xx} Inter-annual variation. Data are the mean values from one stream for 1994 (n=144); 1998 (n=126) and 1999 (n=30). (extracted from Vidal-Abarca et al., 2004).

Diel fluctuations in the physico-chemical parameters are also high in temporary streams because of fluctuating surface water discharge. In an arid Mediterranean stream (southeast Spain), Vidal-Abarca et al. (2002) found that the diel variations for water temperature, dissolved oxygen and pH were higher than those for the annual cycle. However, salinity, alkalinity and nutrient concentrations remained in the annual cycle. This is another important aspect to consider when establishing the natural range of variation of solute concentrations in temporary streams.

On the other hand, we must to be careful when considering expectations on seasonal changes in solute concentrations. On an annual scale, Ortega et al (1988) and García (2005) noted a drop in nitrate-N and chloride concentrations after rainfalls in relation to the base-flow conditions, whereas they observed an increase in the summer months. These results contradict those reported by others authors (e.g., Grimm, 1987; Dent et al, 2001; Martín et al., 2004) who described increased N concentrations after rainfalls. This pattern of variation in N concentrations has been termed by some authors as the “classic N cycle” (Martín et al., 2004).

Likewise, inter-annual variability is another very important aspect to consider, especially in temporary streams, as pointed out by several authors (Martín et al., 2004; Vidal-Abarca and Suárez, 2007; von Schiller et al., 2008a). Shorter-term variability in chemical concentrations in streams and rivers can result from individual events, whereas longer-term seasonal and inter-annual variability are both significant, and provide an indication of environmental changes. The presence of trends in water chemistry provides insight into the contributing factors behind these changes, such as climatic variation or changes in land use and management (Potts et al., 2003).

Finally, Table 4 summarises the spatial and temporal variability data in some physico-chemical variables extracted from different studies conducted in temporary streams of southeast Spain. Just as spatial and temporal variability increases as the study scale does, the study of a broader range of temporary streams from other regions is fundamental to gain a global view of the wide-ranging variation of the physico-chemical features in temporary streams.

In summary, all of these results highlight the importance of considering spatial and temporal variation in solute concentrations and chemical conditions to better design monitoring programme that adequately characterise temporary streams. Although this last aspect may have been somewhat neglected, it is still an important consideration in stream water monitoring programs. In addition, the European Water Framework Directive (WFD)’s ultimate purpose is to determine the ecological and chemical statuses of surface waters, which is another important reason to obtain more information on spatial and temporal variability in temporary streams under chemical conditions.

In addition, the range of natural variation in temporary streams should be incorporated into the quality standards for RC in surface waters, as other authors have indicated for macroinvertebrate assemblages in Mediterranean streams (Sánchez-Montoya et al., 2007).

4.3. The Use of Functional Indicators

The WFD stipulates (European Commission, 2000) that the ecological quality of surface waters should be quantified as “an expression of the quality of the structure and functioning of aquatic ecosystems associated with surface waters”, where the term structure refers to the physical and chemical settings and to the biological structure of ecosystems. The idea assumed by this directive, that measures of biological structure relate directly to ecosystem functioning, is too simple (structure and functioning might not actually be linked) and complex given the fact that linking structural properties of the community to ecosystem processes is a major challenge in contemporary ecology (Sandin and Solimini, 2009). Young et al. (2008) suggested that stressors may change the structure but not the function, both the structure and function, or the function but not the structure. Thus, recently ecosystem functioning has been suggested as a measure of ecosystem integrity and health to complement ecosystem structures (through biological and physico-chemical indicators) that are usually measured during routine monitoring (Gessner and Chauvet, 2002; Fellows et al., 2006). Ecosystem functioning includes a considerable number of relationships (processes) among the ecosystem's biotic and abiotic elements, including those with adjacent ecosystems (terrestrial-aquatic linkages), and it is expected to accurately reflect a broad range of catchments disturbances (Bunn et al., 1999). The importance of these ecosystem processes are becoming increasingly recognised by the WFD. In fact, today's management programmes recognise the importance of protecting ecosystem processes as another ecosystem biodiversity element.

The more commonly used measures of ecosystem functioning are leaf litter breakdown rates, ecosystem metabolism (i.e., gross primary production; GPP and ecosystem respiration; ER) or nutrient retention (nutrient uptake velocity), as well as biofilm biomass, growth chlorophyll *a* concentration, the autotrophic index (biofilm ash-free dry mass/chlorophyll *a*) and fungal biomass (i.e., Feio et al., 2010). The ecological importance of functional indicators has inspired numerous empirical studies on its rates and controls, albeit mostly in a single stream or among a few streams in a single region (e.g., Mulholland et al., 2001; Dodds et al., 2002; Kemp and Dodss, 2002; Gücker and Boëchat, 2004; Wiegner et al., 2005; Bernot et al., 2006; Roberts et al., 2007), and more recently across geographic regions (Bernot et al., 2010). However, the number of studies carried out in temporary streams is much lower, and include those on nutrient retention (Maltchik et al., 1994; Martí and Sabater, 1996; Martí et al., 1997; Butturini and Sabater, 1998; von Schiller et al., 2008a) and metabolism (Mollá et al., 1994; Guasch and Sabater, 1995; Guasch et al., 1995; Mollá et al., 1996; Suárez and Vidal-Abarca, 2000; Velasco et al., 2003; Acuña et al., 2004; 2005).

Likewise in recent years, a large number of studies on land use effects on stream functional indicators have been conducted. Save a few studies carried out in the Mediterranean region (Sabater et al., 2000; Martí et al., 2004; Martí et al., 2006; von Schiller et al., 2007; 2008b; Gutiérrez-Cánovas et al., 2009), the analyses of the influence of anthropogenic stressors on stream ecosystem functioning have practically been limited to temperate regions of North America (e.g., Haggard et al., 2005; Meyer et al., 2005; Houser et al., 2005; Johnson et al., 2009; Bernot et al., 2010). However, studying the influence of stressors on stream function across landscapes and climates (geographic regions) is critical to develop a global view of how land use affects the health and integrity of streams (Bernot et al., 2010). A more profound study of Mediterranean temporary streams would improve knowledge on the effect of land use changes on stream functioning given the specific features characterising Mediterranean streams (von Schiller et al., 2008b): natural deficit of water resources with marked hydrological fluctuations (including drought), a longer history of human impact than temperate streams, and watersheds with mixed land uses that conform a very patchy landscape.

As we can find a wide range of hydrologically different streams under the term Mediterranean, several critical questions may be better addressed in temporary Mediterranean streams before using functional indicators. For instance, how low and intermittent surface flow affects ecosystem processes such as nutrient uptake, ecosystem metabolism or organic matter breakdown; how drought affects all these processes; and the effect of salinity on them.

Many of the aspects highlighted in the previous section can be applied to help provide a better understanding of why a wide spatial and temporal variability in the functional processes in Mediterranean temporary streams can be expected. Physico-chemical factors (nitrate-N and ammonium-N availability, dissolved oxygen concentration, pH, etc.) and the biotic environment determine the rate and reactions of ecosystem processes (MacClain et al., 2003). Similarly,, ecosystem processes (organic matter breakdown, N-nitrification, denitrification, and in general nutrient uptake measures, GPP, ecosystem respiration) affect the physico-chemical environment. Both are linked but describe different aspects of the same entity. Thus, is not unexpected, as suggested for the case of solute concentrations, that spatial and seasonal variability in ecosystem processes will be higher in intermittent streams than in permanent streams.

In several streams located in a single Mediterranean catchment, von Schiller et al. (2008b) found that NH_4^- demand (measured as uptake velocity V*f*) decreased along the gradient from forested-to-urban-dominated watershed, primarily in response to increases in the stream water nutrient concentration. However, our results of NO_3^+ and NH_4^- demand (V*f*), obtained in two Mediterranean temporary streams (SE Spain), differed through a similar gradient of land use change, but through the same channel (i.e., from head to mouth) (Arce and Gómez, *in preparation*). NO_3^+ and NH_4^- demand did not decrease through the stream channel, with impacted reaches (down-reaches) showing similar N demand to un-impacted reaches (up-reaches). In fact in one of the studied streams, NH_4^- demand even increased in impacted reaches if compared with non-impacted reaches. In addition, NO_3^+ uptake showed no sign of saturation (O´Brien and Dodds, 2010), which could be expected if we consider the high NO_3^+ concentrations of these down-reaches (average values of 3-21mg/L) the high concentrations of N added (50 mg/l), and the "expected vulnerability" of temporary streams to N inputs given their scarce flow and limited biota (Dahm et al., 2003; von Schiller et al., 2008a; Gómez et al., 2009).

Hydrological intermittency may have a strong influence on both ecosystem processes and nutrient dynamics (Dahm et al., 2003; Bernal et al., 2005; Lillebo et al., 2007). In particular, biotic uptake (nutrient uptake length) may be influenced by changes in discharge (Stream Solute Workshop, 1990; Peterson et al., 2001; Wollheim et al., 2001), as reported in Mediterranean streams (Martí and Sabater, 1996; von Schiller et al., 2008a). However, the effect of drought on ecosystem processes is not well-known.

By way of example, the effects of drought on N retention in streams and rivers are less thoroughly documented than effects of floods (Kern et al., 1996; Koschorreck, 2005). Although some studies have examined the effect of drought and rewetting on N cycling (N-mineralisation, denitrification) (e.g., Qiu and Mc Comb, 1996; McIntyre et al., 2009a; 2009b; Fierer and Schimel, 2002), there are no studies available on the effect of drought on N uptake, or metabolism. In this sense, some authors (e.g., Amalfitano et al., 2007) have suggested an inherent resistance to drying of part of the benthic bacterial community in Mediterranean temporary rivers, which may better adapt to frequent drying-rewetting events if compared to stream bed microorganisms that rarely undergo extreme fluctuations in moisture content

(Fierer and Schimel, 2002). Similarly, von Schiller et al (2008a) revealed how seasonal variability in nutrient retention in a intermittent stream was unexpectedly lower than in a permanent stream, which suggests the high resilience of the biological communities responsible for nutrient uptake.

Another poorly studied aspects of Mediterranean temporary streams is to know how salinity affects ecosystem functioning. In the Mediterranean region, one of the human stressors on aquatic systems is the dilution of its natural salinity content. The expansion of irrigated surface land has resulted in increasing dilution stress that affects the biological structure of streams (Gómez et al, 2005; Velasco et al., 2006). Along these lines, Gutiérrez-Cánovas et al. (2009) analysed the effect of dilution stress on the functioning of a saline Mediterranean stream, and showed that although the metabolic rates and biomass of consumers were greater under disturbance conditions (dilution), no significant differences between disturbance or non-disturbance conditions were found given their strong dependence on the seasonal cycle. However, autotrophic biomass was clearly affected, with changes in community species composition under the impacted conditions.

These results support the idea that more studies are needed in temporary streams to provide further knowledge of ecosystem functioning and the effect of anthrophogenic stressors on them. This necessity becomes even greater if we consider that on a global scale, more streams and rivers have being affected in its hydrological conditions by natural or human-induced changes. In parallel to surface discharge reduction, increased water salinity occurs as a result of lower dilution. This fact is expected to affect an increasing number of streams and rivers in different regions as surface discharge will decrease. Thus, the effect of salinity on stream ecosystem functioning could be considered a research priority in temporary Mediterranean streams.

Finally by considering the unique properties of temporary streams (Larned et al., 2010), and the still scarcity of functional studies (in relation to driver factors and the effect of stressors), we finish with a call for conservation, resource management and research investment efforts that address their preservation. Studying temporary streams allows us to learn more about their functioning under fluctuating hydrological conditions, a situation which is likely to occur given the expected global climatic change. As Larned et al. (2010) pointed out, the primary objectives for effective temporary river management are the preservation or restoration of aquatic-terrestrial habitat mosaics and natural flow intermittence, and the identification of flow requirements for highly valued species and processes.

CONCLUSIONS

Many studies carried out in Mediterranean streams have reported useful classifications of stream-types in this area, however fewer works have validated the needed correspondence between environmental typology and biological communities. Most of typologies agreement with the division of temporary and permanent in different stream types due to their different biological communities. Lower values for several metrics as EPT, NFAM and IBMWP in temporary streams, if compared to permanent streams, must be attributed to the natural differences in macroinvertebrate communities.

Also a specific methodology to select reference sites in Mediterranean streams has been developed. Its application indicates that, due to the multiple stressors present in Mediterranean areas, not reference sites are found at “large watercourses” which are located at the main course at low altitude. In this case, less restrictive criteria must be established.

Recent studies point out the wide variability in biological communities in temporary streams. This may justify a further division of this ecotype into subgroups to ensure the proper application of the WFD methodology. This subdivision may relate to the varying permanence of superficial flows in intermittent and ephemeral streams and natural salinity.

To consider spatial and temporal variation in solute concentrations and chemical conditions, is critical for designing better monitoring programs to adequately characterize temporary streams and to determine the ecological and chemical status. In addition, the range of natural variation should be incorporated into the quality standards for reference conditions in surface waters.

Functional variables may be especially useful in situations where there is a stronger response among those organisms not usually included in stream assessments (e.g., fungi and bacteria) than the commonly used invertebrates, macrophytes and fish indicators; to detect early signs of degradation in high quality sites. However, we believe that more studies must be carried out, especially in temporary Mediterranean streams, to obtain functional indicators to be used as an effective, applicable tool for streams and rivers management.

Finally, in the framework of the global climatic change predictions, studies conducted in temporary streams are essential to make predictions on the spatial and temporal variability in the structure and functioning of the aquatic ecosystems at a wider global scale.

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Chapter 3

HISTORICAL CHANGE AND MANAGEMENT OF THE FLOODPLAIN FORESTS OF THE MIDDLE ELBE RIVER

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ABSTRACT

Floodplain forests are vital to the functioning of river ecosystems. They support the rich diversity characteristic of these systems, play an important role in nutrient cycling, and serve as an ecotone between the terrestrial and aquatic biomes. This chapter explores the changes that have taken place in the floodplain forest community of the Middle Elbe River over the last 50 years, examining the role that altered river dynamics, herbivore pressure, human management, and invasive species have played in shaping the current forests. It also looks at the development of the UNESCO Flusslandschaft Elbe Biosphere Reserve, which now manages much of the Middle Elbe's ecosystem, and the influence the biosphere reserve's activities have on the forests. Our research indicates that the floodplain forests, currently dominated by pedunculate oak (*Quercus robur)* and field maple (*Acer campestre*), have undergone a shift in community composition since the 1950’s, including a loss of more flood tolerant species. Our results further suggest that browsing is limiting regeneration. When examining the forests across the different management zones, it appears that the core zone forests will likely experience an eventual loss of oak dominance, whereas forestry practices in the buffer zones should maintain the oak at a moderate level. American green ash (*Fraxinus pennsylvanica*), an introduced species, is also more prevalent in the buffer zones. The different zones of the biosphere reserve currently help maintain a more diverse mix of forest communities across the landscape. However, without a restoration of a more natural flooding regime, early successional communities will become increasingly rare. The biosphere reserve managers recognize this issue and are currently pursuing several projects to help mitigate the loss of natural processes, including setting back levees to increase the size of the active floodplain, restoring backwater lakes, and removing introduced species

INTRODUCTION

Natural river floodplain forest systems in Europe have almost completely been destroyed by extensive modification of European rivers and their floodplains (BRFME2002). Remnant systems are valuable ecologically, as well as socially. In general, they represent some of the most diverse habitats in the temperate zones, are highly productive, and provide a multitude of services, such as water filtration, flood control, recreation, wildlife habitat, and migration corridors (Petts 1990; Pinay and others 1991; Ward and Stanford. 1995; Décamps 1996; Dister 1996; Costanza and others 1997). Some of the few remaining systems in Europe have begun to receive legal protections, e.g. the Lower Oder National Park, the Alluvial Zone National Park on the Danube, and the Flusslandschaft Elbe Biosphere Reserve. Despite protections, remnant systems face a multitude of threats, including the loss of the natural flooding regime, logging, introduced disease and species, and altered levels of herbivory (Dynesius and Nilsson 1994; Dister 1996; Scherzinger 1996; Tickner and others 2001). To conserve and protect these and other remnant areas properly, we must better understand how such systems function and how they respond to potential threats. This chapter reviews basic floodplain forest ecology before examining the floodplain forests of the Middle Elbe and the changes that have occurred there over the last fifty years. Additionally describes the development of the Flusslandschaft Elbe Biosphere Reserve and analyzes the impact the biosphere reserve model has had on the forests. Lastly, it presents some exemplary management practices the biosphere reserve implements to maintain the health of the forests.

Temperate Floodplain Forests

River floodplain forests are known in the literature as alluvial forests, bottomland hardwood forests, or riparian forests. They represent some of the most diverse and productive systems on this planet, but also some of the most threatened (Brinson 1990; Malanson 1993; Christensen and others 1996; Décamps 1996; Ward 1999). The natural hydrologic regime, characterized by periodic flood-pulses, is the major factor shaping natural floodplain forests. The floodplains often include natural levees along the banks of a river, behind which a ridge-swale topography dominates—the result of historical shifts in the river channel (Brinson 1990). The small variations in elevation above the river translate into large differences in the frequency and duration of flooding at a given location. This, in turn, greatly influences the type of vegetation that can grow there. Flood-tolerant species compete well at the lower elevations, where flooding events are more common and of longer duration. Species with less flood tolerance are restricted to higher elevations sites, such as ridges, levees, and terraces. (Brinson 1990; Naiman and Décamps 1997). The deposition of sediments with the flood pulses can slowly increase the elevation of a site above the river and groundwater, allowing succession from more to less flood-tolerant species (Barnes 1997). Thus, in contrast to many upland forests, where light availability and shade tolerance often drive forest ecology (Oliver and Larson 1996; Scherzinger 1996), floodplain forest composition and succession are determined primarily by flooding. Hall and Harcombe (1998) demonstrated that flooding can even alter a species' shade tolerance. However, succession is not unidirectional in these systems; extreme flooding events can reset community development on the floodplain (Junk and others 1989; Naiman and Décamps 1997; Yin 1998). The net result of these dynamic processes is a diverse mix of species and communities for which floodplain forests are well-known.

Functions and Values of the Floodplain Forests

Floodplain forests contribute greatly to the ecological health and the functioning of the entire river-floodplain system (Décamps 1996). The diverse flora provides habitat and forage for numerous species of insects, reptiles, amphibians, birds, and mammals; during flood periods, they also play the same role for many species of fish and other aquatic organisms (Petts 1990; Gore and Shields Jr. 1995; Piegay and Landon 1997). During difficult periods, such as regional drought, these forests can serve as refugia (Ware 1955; Naiman and Décamps 1997). They also serve as seasonal migration corridors, most notably for migratory bird species (Junk and others 1989; Petts 1990; Naiman and Décamps 1997). Floodplains also play an important role in the storage of floodwaters, as well as the recharge of local groundwater (Pinay and others 1991). The forests also filter surface and shallow groundwater and contribute to nutrient cycling while water is stored on or as it flows through the floodplain. They moderate levels of nutrients such as nitrogen and phosphorus arriving from uplands and upstream, as well as trap sediments running off from upland sources (Peterjohn and Correll 1984; Pinay and others 1991; Lowrance 1999). The forests also serve as a source of a variety of forms of organic carbon, which provides food for river organisms and in its largest form (large woody debris) also traps sediments (Petts 1990; Ward and Stanford. 1995; Décamps 1996; Piegay and Landon 1997).

These functions are not just important from an ecological perspective, but they are also services for human societies (Costanza and others 1997; Ewel 1997; Postel and Carpenter 1997). The diversity of plants and animals in these areas provides economic and cultural benefits, such as bird watching, fishing, hunting, and timber. In areas where runoff from farms and settlements often contain excessive levels of nutrients and sediments, the filtering functions of floodplain forests help counteract cultural eutrophication of receiving water bodies. Furthermore, the floodwater storage capacity of these forests is important for protecting downstream settlements in floodplain areas from extreme flooding events. The lack of active floodplains along the Elbe created dangerous flooding situations for human settlements, which resulted in significant loss of human life and property during the 2002 floods (WWF Deutschland 2009). Finally, where communities draw water from riverine aquifers, the groundwater-recharge function of natural floodplain forests is invaluable.

Human Interventions in River-Floodplain Forests

Human activities have increasingly affected temperate floodplains. Tockner and Stanford (2002) estimate that in North America (excluding Alaska and N. Canada) approximately 46% of the floodplain is "intensely cultivated"; in Europe, the corresponding figure is 79%. Humans have become significant forces shaping the floodplain system and detrimental ones with respect to the ecological health of the forests. In North America, over half of the forested wetlands have been destroyed; in Europe, natural floodplain losses approach 95% (Tockner and Stanford 2002).

The major impacts stem from engineering works in the river and on the floodplain, such as dams and levees. Forest conversion, timber harvests, introduced diseases, invasive species, and altered levels of herbivory also threaten the forests. Successful conservation of remnant floodplain forests must concern itself with these potential threats.

River engineering works disrupt the natural flow regime of river systems and their floodplains. Dams moderate natural high and low flow events downstream, thus depriving floodplain forests of their most important disturbance (Décamps and others 1988; Sparks 1995; Poff and others 1997). Dams also trap sediments, which causes the river downstream to incise into its bed (Brinson 1990; Dister 1996; Poff and others 1997). This lowers the elevation of the river with respect to the floodplain and drops groundwater levels. Both of these impacts further exacerbate the decrease in flooding caused by upstream water retention. Wing dams and other river training works also contribute to bed erosion. Levees, in contrast, prevent floodwaters from inundating those parts of the floodplain that lie behind the levees (Gergel 2002).

All of these alterations modify the rejuvenation of the existing forest community by either decreasing or eliminating periodic floods. Consequently, the forest community shifts to later successional stages, losing pioneer stages and diversity (Johnson and others 1976; Barnes 1997; Poff and others 1997). The changes in the hydrologic regime can render these systems more susceptible to invasive species as well (Poff and others 1997; Cowell and Dyer 2002). These changes in the vegetative community can then cascade through the wildlife of the region, further stressing already threatened species (Naiman and Décamps 1997; Knutson and Klaas 1998; Nolet and Rosell 1998).

Timber harvests in the floodplain may provide economic benefits to surrounding communities (Petts 1990; Ewel 1997), but they also represent disturbances that create gaps in the forest and increase the amount of light reaching the lower layers. The exact nature of the impact from a harvest depends on the type of harvest (Lockaby and others 1997). Clearcuts typically have the greatest effect on species composition and shift regeneration toward shade intolerant species (Dister 1996; Scherzinger 1996; Meadows and Stanturf 1997; Messina and others 1997). Harvests such as group and single-tree selection tend to favor more shade tolerant species. Trees that remain on site after these partial harvests sometimes suffer from either direct damage from harvests or indirect impacts from increased exposure to light or to greater wind intensities (Meadows and Stanturf 1997; Kozlowski 2002). Although harvests provide a disturbance that may enhance the reproduction of shade intolerant species, they also provide an opportunity that invasive species can exploit (Knutson and Klaas 1998). Harvests can also, at least temporarily, decrease primary productivity, evapotranspiration, nutrient cycling, and the filtering of overland flow (Brinson 1990; Shepard 1994; Lockaby and others 1997).

Introduced diseases and species present another set of threats to the floodplain forests. Dutch elm disease, a fungal disease native to Asia, has decimated major elm species in Europe and North America over the last century.

One of the most affected communities has been the temperate floodplain forests, where elm species were historically dominant canopy species (Dunn 1986; Wagner 2000). Since the 1920's, Dutch elm disease has affected elm populations in Europe, particularly the small-leaved elm (*Ulmus minor*), which historically has been a major species in German river floodplain forests (Scherzinger 1996; Schütt and others 2002). Elms infected with the disease typically die before they reach 40 years, preventing them from attaining a dominant position in canopy (Dornbusch 1988). The gaps created by dead elms can lead to increased shrub density in the understory (Dunn 1986).

Invasive species are also a growing concern in floodplain forests in the temperate zones (Tickner and others 2001) . Invasive species can typically take advantage of the altered disturbance regimes of the floodplain forests more readily than many of the native species (Planty-Tabacchi and others 1995).

Invasive species can potentially alter the structure and composition of forests, as well as to reduce regeneration of native species. In their review of exotic and invasive species trends, Schnitzler and others (2007) found several exotic tree species present in European riparian forests; these included box elder (*Acer negundo*), Tree of Heaven (*Ailanthus altissima*), green ash (*Fraxinus pennsylvanica*), honey locust (*Gleditschia triacanthos*), and black locust (*Robinia pseudoacacia*).

Herbivory is also considered one of the major factors shaping natural communities (Naiman and Décamps 1997). Over the last several centuries, North American and Western European societies have systematically extirpated major predators and encouraged large herbivore populations through land cover changes and hunting practices (Scherzinger 1996; Waller and Alverson 1997). Human activities have led to a large increase in herbivore populations in German forests, in particular the roe deer, *Capreolus capreolus* (Briedermann and others 1986; Schwartz 1988; Scherzinger 1996).

Studies have shown that deer browsing can hinder tree regeneration, alter forest composition (Gill and Beardall 2001; Harmer 2001; Partl and others 2002), and at least one study demonstrates that browsing tolerance is more important than shade tolerance in forests with excessively high deer densities (Collins and Carson 2003).

The Elbe and the Middle Elbe

As seen in Figure 1, the Elbe flows 1094 km from its source in the Giant Mountains (Krkonoše) of the Czech Republic to Cuxhaven, where it enters the North Sea with an annual average discharge of 861 m^3/s (Simon and others 2005). The Elbe catchment measures 148,248 km^2, with two-thirds in Germany, one third in the Czech Republic, and some very small areas in Poland and Austria (International Commission for the Protection of the Elbe 2005). The river is typically divided into three main sections: the Upper Elbe, which runs from the source to Schloss Hirschstein in Germany; the Middle Elbe, which continues until the Geesthacht Weir; and the Lower (tidal) Elbe, which completes the course to the North Sea at Cuxhaven.

The Elbe has not been channelized and remains relatively undeveloped, compared to other major European rivers, such as the Rhine and Danube (Reichhoff 1991a). Still, existing islands and sand bars were removed at the turn of the 19th century, and many meanders were shortened (Jüngel 1993); it is estimated that the entire Elbe has lost 119 km since the 16th century (Simon and others 2005).

The river also possesses wing dams and other channel training structures, many of which fell into disrepair during the Cold War (Dörfler 1996). Although the Middle Elbe is free flowing, its flow regime has been affected by the multitude of dams in the upper watershed, which have at least partially been responsible for a decrease in flooding during the second half of the 20th century (van der Ploeg and Schweigert 2001; Helms and others 2002) and an increase in the level of low flows (Simon and others 2005). Levees were first built as early as the 1100's by settlers to the Middle Elbe. Today 730 km of levees line the Middle Elbe, preventing floods from reaching 3285 km^2 (approximately 80%) of formerly active floodplains (Bräuer and Lozán 1996; Simon and others 2005).

The Middle Elbe possesses a temperate climate with an annual temperature of 8.7 °C, ranging from 0°C in the winter to 18.5°C in the summer (Ständige Arbeitsgruppe der Biosphärenreservate in Deutschland 1995); average annual precipitation is approximately 550 mm. At Aken, the discharge rate averages 431 m^3/s (Bergemann 2006). The highest flows along the Middle Elbe typically occur in springtime, with an average high flow of 1650 m^3/s (Simon and others 2005). Flows decrease throughout the summer and fall, reaching an average low flow of 158 m^3/s. Nonetheless, floods can occur throughout the year due to snow melt, heavy precipitation events, and/or ice jams.

The forests of the middle Elbe represent the largest remaining floodplain forest complex of Central Europe (Reichhoff 1991b). They are typically classified under the Braun-Blaunquet system (Braun-Blanquet 1965) with two main associations: the *Weichholzaue* (Salici-Populetum), which represents the pioneer community of multiple willow (*Salix* spp.) and poplar (*Populus* spp.) species; and the *Hartholzaue* (Fraxino-Ulmetum Subass. Acer campestre), which is dominated by four species: pedunculate oak (*Quercus robur*), the small-leaved elm, the common European ash *(Fraxinus excelsior*), and the field maple (*Acer campestre*).

On drier sites, hornbeam (*Carpinus betulus*) and linden (*Tilia cordata*) are also present. Although European floodplain forests have fewer species than their North American counterparts, they are still considered the most diverse forests in Europe (Schnitzler and others 2003). Further, they possess a high level of biodiversity: more than 40 mammal species, 35 fish species, and over 280 bird species have been recorded in the Middle Elbe region (Dornbusch and Dornbusch 1991; Adams and others 2001)

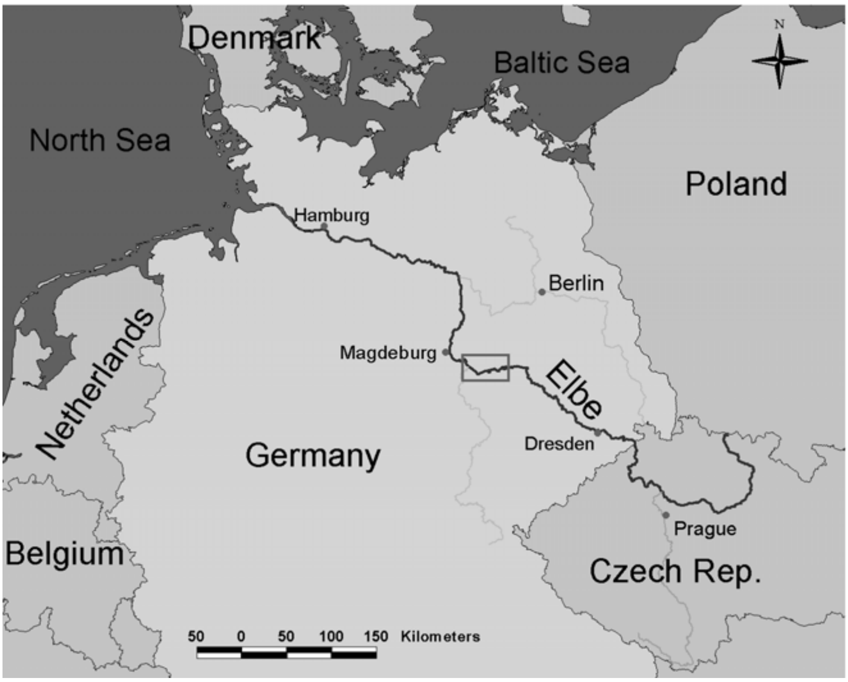

Figure 1.The Elbe River.

FOREST CHANGE SINCE THE 1950'S

The co-existence of high levels of human activity along the Middle Elbe with the remnant forest complexes raises important questions about how these forests are responding to the human pressures. This section will first look at how the forests have changed since the 1950's and then consider the role the development of the biosphere reserve has played.

To examine how the floodplain forests have changed since the 1950's, we selected sites in the active floodplain, not behind the levee system (Boxed area in Figure 1. and Table 1) and in areas studied by Passarge (1953, 1956). We sampled each site using the random pairs method, a plotless survey technique developed by Cottam and Curtis (1956), which provides quantitative data on forest structure and composition. We collected data at sampling points along randomly located transects that ran perpendicular to the flow of the river. To avoid edge effects, we did not sample the 25 m near the forest edge. We also did not sample in areas that had been recently harvested, as they lacked an overstory and only consisted of planted mixes of oak, ash, and elm. At each point, we collected data on species, diameter at basal height (dbh), and distance between pairs of individuals for both the tree and understory layers. These data allow for the estimation of frequency, stem density, and basal area (tree layer) for the sites. The tree layer consisted of stems with a dbh greater than 10 cm and the understory as stems taller than 1.5 m and with a dbh of 1-10 cm. We treated individuals with multi-stems as one entity. In all, we sampled forty points per site.

Table 1. Study sites

Site	Management	Year[a]	Management comments[b]

	Zone		
East 1	core	1961(90)	Stands range in age from 85 – 165 years; some trees over 250 yrs.
West 1	core	1955(79/90)	Stands range from 40 to over 220 years
West 2	core	1961(79/90)	Stands range from 40 to 220 yrs.
East 2	buffer	1961	Stands range from 30 – 135 yrs
East 3	buffer	1961	Stands range from 45 – 135 yrs.
West 3	buffer	1926	Stands approximately 80 years old; some trees 150-200 yrs[c].

[a] year established as *Naturschutzgebiet* (conservation reserve); year established as core zone in parentheses.
[b] source: Haideburg, Lödderitz, and Wörlitz Forestry Bureaus
[c] data were not available for all of study site

Historical Data

For the historical analysis, we compared our data with those of Passarge (1953; 1956). The 1953 study provides data on the floodplain forests in the so-called Middle German Dry Area (the western half of the MEBR) and the 1956 study provides data on the forests of the eastern half of the MEBR. Passarge collected data using methods typical of German ecologists, the Braun-Blanquet School. This methodology differs from the methodology of the Wisconsin School we used and provides data in a different, less quantitative form. Site selection is generally subjective, as the investigator typically chooses a representative area for the community of interest (Kreeb 1983). It uses one large quadrat for forested areas (typically 400 m^2) per relevé and collects data on a species (visually) estimated cover. Cover estimates are given in categories (e.g. +, 1-5). Multiple relevés analyzed provide a measure of frequency (Stetigkeit).

Data Comparison And Analysis

For analysis purposes, we transformed the data from the tables in Passarge's studies into importance values for the species. The studies provided data on both cover and frequency. We calculated average cover values where necessary, and then converted the cover and frequency values into relative cover and frequency, respectively. We summed the relative cover and frequency to produce an importance value. We averaged IV's from both the 1953 and 1956 studies. To create comparable IV's from our data, we assumed relative dominance (basal area) for the tree layer and relative density for the understory to be proxies to Passarge's relative cover. We calculated frequency and relative frequency based on site-level data, rather than point-level data, as would be typical with the Wisconsin methods. We summed the appropriate relative abundance measure with relative frequency for species to yield an IV.

For our data, we calculated summary data statistics based on site-level data, including average basal area per tree, basal area per hectare, tree stem density, and understory stem density. Since Passarge's studies do not contain comparable information, we provide estimates from Dornbusch (1988), which contains basal area and density estimates for the Steckby-Lödderitz forest from 1971. For each time period, we also calculated several measures of diversity for both the tree layer and the understory: number of species, the Shannon index (H'), and Pielou's J, a measure of evenness (Ludwig and Reynolds 1988).

We also calculated IV's for groups of species based on life-history traits and other traits of interest. These include shade tolerance, flood tolerance, two measures of ecological quality, and a measure of herbivory. Appendix 1 provides a summary of the species and their respective classifications. We divided species into three classes of shade based on available literature: tolerant, intermediate, and intolerant (Schnitzler and others 1992; Wagner 1995; Grubb and others 1996; Niinemets and Kull 1998; Kollmann and Grubb 2001; Atkinson and Atkinson 2002; Schütt and others 2002; Tallantire 2002). We divided species into similar groupings of flood tolerance, again based on the available literature (Schnelle 1981; Dister 1984; Frye and Grosse 1992; Schnitzler and others 1992; Siebel and Bouwma 1998; Rothmaler 1999; Atkinson and Atkinson 2002; Schütt and others 2002; Tallantire 2002). One measure of ecological quality derived from a work by Hermy and others. (1999), which analyzed ancient forest remnants in Europe and determined a group of "ancient forest species," species common in forests that are several hundred years old. The other measure of ecological quality derived from "Hemerobie" rankings, a measure of a species' preference for human disturbed habitats (Bundesamt für Naturschutz 2002). Based on these ranking, we classified a species as more common in relatively natural (high conservation), moderately disturbed (intermediate), or disturbed habitats (low conservation), depending on the average hemerobie value. For an indicator of herbivory by roe deer, we classified plant species as palatable, intermediate, or unpalatable, based on the literature (Bobek and others 1979; Gill 1992; Verheyden-Tixier and others 1998; Gill 2000; Gill and Beardall 2001; Harmer 2001; Atkinson and Atkinson 2002; Partl and others 2002; Schütt and others 2002). Finally, for the understory, we also examined changes in a grouping of species armed with thorns or other herbivore defenses, as another potential indicator of herbivory (Rothmaler 1999; Schütt and others 2002).

Changes in the Tree Layer

The forests of the Middle Elbe have been heavily managed for centuries (Wagner 2000; Schmidt 2002). In the last couple of decades, management has shifted its emphasis toward using and restoring natural processes and species. The importance values for the species in the tree layer demonstrate that some large changes have taken place (Table 2). Four species dominated the 1950's forests: European ash, pedunculate oak, small-leaved elm, and field maple. In contrast, pedunculate oak strongly dominates today's forests, although European ash and field maple remain somewhat dominant despite large decreases in their importance.

The small-leaved elm has decreased greatly in importance and no longer rates as a dominant species. Although the oak is not particularly shade tolerant, it is flood and drought tolerant (Dister 1984; Schütt and others 2002). Thus, it is very competitive in the floodplain forests once established. Oak's increase in dominance relates not only to its own life history characteristics, but also to the loss of the small-leaved elm from Dutch elm disease and possibly to competition between the European and the (introduced) green ash. The other common elm species, European white elm (*U. laevis*), has remained relatively constant. There has been a large increase in the importance of the introduced American green ash, as well as hybrid poplars and horse chestnut (*Aesculus hippocastanum*). It is also interesting to note that the overall importance of the genus *Fraxinus* has not changed. Dominant species were not the only species showing changes. Wild apple (*Malus sylvestris*) and bird cherry (*Prunus padus*) have both experienced decreases. These species are both typical floodplain trees and not especially shade tolerant (Bundesamt für Naturschutz 2002; Schütt and others 2002). Thus, they would be less competitive in the current floodplain where flooding disturbances have decreased in magnitude and frequency. On the other hand, sycamore maple, which is shade tolerant but less flood-tolerant, has increased (Bundesamt für Naturschutz 2002). We were unable to judge historical changes in several species, as they were absent from Passarge's (1953; 1956) studies; these include green ash, horse chestnut, and the hybrid poplars. A researcher employing the Braun-Blanquet system to investigate the native floodplain forests would have likely avoided sites containing these species. Nonetheless, historical forestry records indicate that all three species were present in the 1950's (Wagner 2000; Schmidt 2002).

The difference in methods explains, at least in part, the differences in the diversity measures from Passarge (1953, 1956) and our data (Table 3). The present forests showed a greater level of diversity than the forests in the 1950's. Excluding non-tree species, the current forests possess seven more species than recorded 50 years ago. The Shannon index is similar for both time periods, but the evenness measure has experienced a large decrease, which relates to the large increase in oak dominance and the mortality of a large number of elms.

Table 2. Importance Values for the Tree Layer: (species with increased values are bolded)

Species	1950's	current
Fraxinus excelsior	41.9	22.3
Quercus robur	41.3	57.3
Ulmus minor	33.8	10.3
Acer campestre	30.7	23.2
Ulmus laevis	14.2	14.1
Carpinus betulus	10.7	10.1
Malus sylvestris	9.6	3.6
Tilia cordata	8.2	4.0
Prunus padus	5.6	1.7
Acer pseudoplantanus	1.8	6.5
Pyrus pyraster	1.7	1.8
Fraxinus pennsylvanica	0.0	17.7
Populus spp.	0.0	7.8
Aesculus hippocastanum	0.0	4.8
Betula pendula	0.0	2.1
Rhamnus cathartica	0.0	2.0
Fraxinus spp.[1]	41.9	39.9

[1] Sum of F. excelsior and F. pennsylvanica

Table 3. Community Composition Indicators for the Floodplain Forests

	Tree		Understory	
	1950's	2000[a]	1950's	2000
Species richness	11	18 (21)	14	19
H'	2.05	2.01 (1.96)	2.11	2.17
J	0.85	0.68 (0.66)	0.80	0.74

[a] Figures in parentheses include all woody species measured with a dbh > 10 cm. The figures outside include only tree species.

Table 4. provides data on several basic measures of forest structure for the current MEBR floodplain forests. Data from Dornbusch (1988) from 1971 indicate that basal area has increased over 30 years, whereas stem densities have likely been stable, although they are difficult to compare directly, given different estimation methodologies.

Table 4. Forest Characteristics of the Middle Elbe Floodplain Forests

Characteristic	Dornbusch[a]	Current
Average basal area per tree (dm^2)	NA	14.9
Basal area per ha ($m^2 * ha^{-1}$)	31.1	36.7
Density (stems * ha^{-1})	514[b]	249
Understory density (stems * ha^{-1})		241 (569[c])

[a] Data are from 1971 and are for Steckby-Loedderitz forest; data are for all woody species greater than 4 cm dbh.

[b] includes all stems of an individual

[c] assumes a conservative average of 5 stems for *Corylus avellana*, *Crataegus* spp., and *Sambucus nigra*.

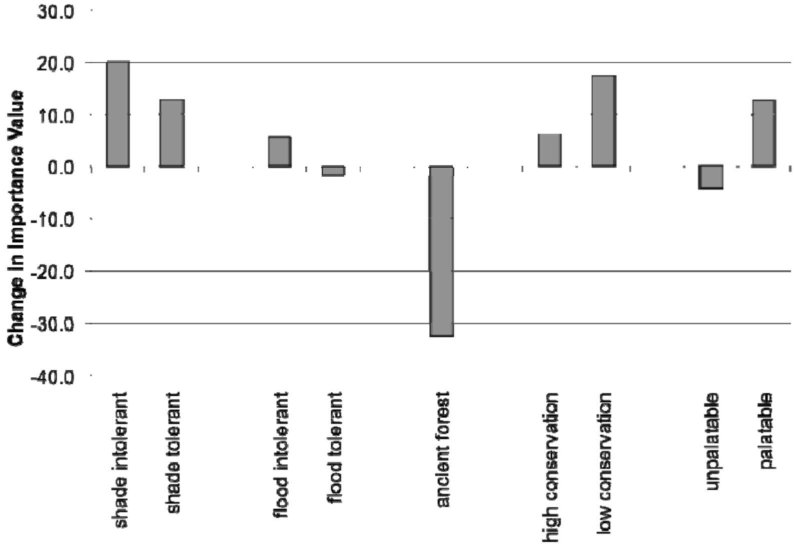

Figure 2. Changes in Selected Life-History Traits of the Tree Layer.

An examination of the life history characteristics of the tree layer also revealed some interesting trends (Figure 2). Despite the changes in the flow regime over the last several decades, we did not detect a major shift in flood tolerance in the tree layer; however, the oak's dominance could mask any potential shifts, due to its flood tolerance. However, its current dominance is more human-driven than ecologically, as oak species were the preferred species in Mittelwald systems (Scherzinger 1996; Wagner 2000). Perhaps a more expected result was the increase in the importance of shade-tolerant species, which indicates a shift toward a later successional stage. As discussed earlier, shade tolerance should become more important as flooding disturbances decrease, canopy disturbances decrease, and competition for light becomes the driving factor in the understory. Our examination of the two biodiversity measures also indicates some deterioration in the quality of the tree layer (i.e. decrease in ancient forest species and increase in low conservation value). This likely relates to Passarge's site selection and his probable avoidance of three species that are lower in conservation value and not considered ancient forest species (green ash, horse chestnut, and hybrid poplars).

Changes in the Understory

Perhaps more indicative of community change are the shifts occurring in the understory. As regeneration is largely allowed to occur naturally, the understory is less influenced by active human management of the forest (though we will discuss an exception below). However, the composition of the understory is also a partial function of the overstory (Messina and others 1997), so historical management continues to influence understory dynamics to some extent. The regeneration trends we see in the understory point to an uncertain future. First, the current density of individuals is low (Table 3). While it is true that typical floodplain forests often demonstrate low understory densities (Jones and others 1994; Kellison and others 1998), the mortality of a significant portion of the canopy in recent decades due to Dutch elm disease should have increased density of the understory (Dunn 1986). Indeed this appears to be the case on other rivers experiencing a large loss of elms, such as the Rhine and the Wisconsin (Schnitzler 1994; Hale and others 2008). Jones and others (1994) mentioned that periodic droughts and increased populations of large herbivores could lower sapling densities—factors, which both apply to the current situation on the Elbe.

Table 5. provides the importance values for the species in both time periods and gives insight into another issue of regeneration. Neither of the dominant species, the small-leaved elm and hawthorn (*Crataegus* spp.) can become canopy species: small-leaved elm, which usually succumbs to Dutch elm disease within 40 years (Dornbusch 1988) and hawthorn, a small tree species at best. In contrast, signs of regeneration of the overstory dominant, pedunculate oak, were absent in our survey, although several saplings were noted outside of the sample area. Thus, barring human intervention, the oak will likely experience a large decrease in importance over the next centuries. Only a few tree species currently possess moderate levels of regeneration: field maple, which can reach considerable sizes in this area (Passarge 1953), and the two ash species. Several species not detected in the 1950's, such as American green ash, silver birch (*Betula pendula*), wild cherry (*Prunus avium*), white elm, and buckthorn, are now present at low levels in the understory. In contrast, several species such as common dogwood (*Cornus sanguinea*), bird cherry (*Prunus padus*), and spindle (*Euonymus europaea*) have all declined.

Table 5. Historical Changes in Understory Importance Values (species with increasing values are bolded)

Species	1950's	current

Ulmus minor	75.7	38.6
Crataegus spp	18.7	36.3
Cornus sanguinea	18.0	11.3
Acer campestre	17.1	20.1
Prunus padus	14.0	2.9
Corylus avellana	13.0	9.4
Euonymus europaea	12.7	2.2
Fraxinus excelsior	9.1	14.0
Prunus spinosa	8.2	7.6
Sambucus nigra	3.7	9.1
Species	1950's	current
Tilia cordata	3.2	7.3
Carpinus betulus	2.5	7.1
Quercus robur	2.3	0.0
Acer pseudoplantanus	1.7	6.8
Fraxinus pennsylvanica	0.0	10.7
Ulmus laevis	0.0	6.0
Betula pendula	0.0	2.4
Prunus avium	0.0	2.4
Rhamnus cathartica	0.0	2.2
Acer platanoides	0.0	1.8
Malus sylvestris	0.0	1.8
Ulmus spp.	75.7	44.6
Fraxinus spp.	9.1	24.7

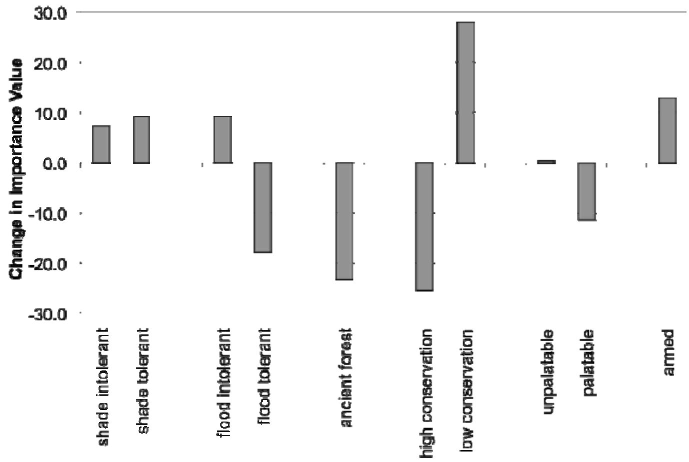

Figure 3. Changes in Selected Life-History Traits of the Understory.

The shift in species traits provides further indication of large shifts in the floodplain forest community (Figure 3). As expected, there was a strong decrease in the number of flood tolerant species with a concurrent increase in both flood intermediate and intolerant species. These results reflect the increases in sycamore maple, hornbeam, and linden regeneration—all relatively flood intolerant—as well as the decrease/absence of oak and bird cherry regeneration, which are both typical floodplain species. This shift in flood tolerance is common on regulated rivers (Schnitzler 1994; Barnes 1997; Knutson and Klaas 1998; Hale and others 2008). Further, these increases also contribute to the large increase in the number of shade tolerant species, which suggests a shift toward a later successional forests (Scherzinger 1996). The diversity indices (Table 3) indicated a similar to increasing levels of biodiversity over the last fifty years. However, it is important to keep in mind that some of this difference relates to the different sampling methodologies. Examining the ancient forest species and the conservation value groupings (Figure 3), we see a different picture. These measures, likely less biased by sampling methods, suggest that biodiversity is lower than it was fifty years ago. The importance of ancient forest species has declined and the importance of lower conservation value species has increased. As with the tree layer, the increase in low conservation species relates in part to the sampling methodology of the data from the 1950's, as green ash is present in the understory. It is also important to note that the ancient forest and conservation value measures reflect human values more so than necessarily ecological ones (Scherzinger 1996). Given the simultaneous use of the forests for wood production, recreation, and preservation, we likely need to accept a compromise between a forest representative of "old-growth" and one more characteristic of anthropogenic disturbance. Figure 3. confirms another important factor shaping the forest—deer browsing. The indices here show a large decrease in the importance of species preferred by roe deer, as well as an increase in species that possess significant deterrents to herbivory, such as thorns. These findings highlight the need to control the roe deer population, which has thus far been a difficult task. Our results also highlight the presence of the introduced green ash, which has become a noteworthy component of the forest. It appears to compete with the native ash species, and perhaps in some cases, has outcompeted it in the past, due to the green ash's greater flood tolerance (Wagner 2000). However, an eradication program could be expensive, require a large effort, and may in the end still be unsuccessful. Furthermore, its removal from core zones would require a change in the "no intervention" policy, which could open the door for other interventions into the core zone forests. WWF Deutschland has also documented the prevalence of green ash in the area of one of its major projects (WWF Deutschland 2010). One of the problems they mention with the spread of green ash is that it could jeopardize the protected status of areas where it has become dominant; both European and German State laws limit the percentage of non-native species in designated protected areas.

MANAGEMENT INFLUENCES

The problems highlighted above in our historical analysis comprise the challenges that the managers of the Flusslandschaft Biosphere Reserve face in the conservation of these important floodplain forest remnants. In the following sections, we discuss the development of the biosphere reserve, its influence on the floodplain forest, and some of the management approaches it is using to deal with the issues.

The Development of the Biosphere Reserve

The Flusslandschaft Elbe (*River-landscape Elbe*) Biosphere Reserve had its origins in a small biosphere reserve established in 1979 in one of the core forest areas, the Steckby-Lödderitz Forest (Dornbusch 1991) located on the Elbe between Dessau and Magdeburg. In 1990, one of the last acts of the German Democratic Republic's legislature expanded this reserve into the Middle Elbe Biosphere Reserve, which spanned 78 km of the Middle Elbe and approximately 25 km of the Mulde and 12 km of the Saale tributaries; it encompassed 43,000 ha, of which 30% was floodplain forest. In 1997, the Middle Elbe Biosphere Reserve was again expanded to the Flusslandschaft Elbe Biosphere Reserve, moving beyond the borders of the German state of Saxony-Anhalt to cross five German states, encompass 400 km of the Elbe and over 370,000 ha of the Elbe's floodplain (BRFME 2009; DUNESCO 2010). UNESCO, through its Man and Biosphere program, has a network of 408 biosphere reserves in 94 countries (Man and Biosphere Programme 2002). The biosphere reserve model is one with general ecological applicability, including river-floodplain systems. Nonetheless, few biosphere reserves worldwide specifically protect temperate river-floodplain forests. As a UNESCO Biosphere Reserve, the Flusslandschaft Elbe's main goals are to protect the biodiversity and ecological functions of the reserve, to protect the cultural resources within the reserve, and to promote the sustainable use of the reserve's resources (Ständige Arbeitsgruppe der Biosphärenreservate in Deutschland 1995). The land in the reserve is a mix of public and private land. All land falls into one of four zones. The *core zone* must be state-owned land and excludes all human uses. The *buffer zone* also must be state-owned, but allows extensive and sustainable uses of its resources. The *transition zone* (or the "zone of the harmonious cultural landscape" in German) can be either state or private land and is where most human use is concentrated. Finally, the *regeneration zone* is degraded land (e.g. former military installations) that is being rehabilitated for eventual reclassification as a buffer or core zone (Ständige Arbeitsgruppe der Biosphärenreservate in Deutschland 1995).

Management Analysis

Due to the relatively young age of most of the River Landscape Elbe Biosphere Reserve, we focus our analysis on the forests of Middle Elbe Biosphere discussed above. We use the data collected above to examine potential differences between the core and buffer management zones. To do so, we pooled data from each site within the respective zone (Table 1) and calculated summary data measures, such as mean basal area per tree, average percent of large tree (dbh > 60 cm) per site, basal area per hectare, tree stem density, and understory stem density. To test for statistical differences in these measures, we used a Wilcoxin signed rank test (SPLUS). We calculated the same measures of diversity for the zones, as we did in the historical analysis. The evaluation of IV's and species traits follows the same method described above. However, since this analysis solely applies to the data we collected, we calculated IV's as follows: for the tree layer, the IV is the sum of relative density and relative dominance (basal area); for the understory, the IV is the sum of relative density and relative frequency. Also, since the MEBR has only been in existence for the past 20 years, we limit our discussion primarily to the understory, which is the layer most likely to show a difference over this time scale (Hall and Harcombe 2001). Our data also permit a rigorous analysis of community composition. We tested for significant differences in community composition of both layers across zones using multi-response permutation procedures (MRPP) in PC-ORD with a rank-transformed Sørensen distance matrix (McCune and Mefford 1999). MRPP is a statistical technique to assess significant differences in the composition of two or more groups

(Biondini and others 1985; McCune and Grace 2002). As a non-parametric method, it does not rely on the assumption of a normal distribution, which ecological data often do not possess. Finally, we performed an indicator species analysis in PC-ORD using both species and trait groupings to identify potential indicators for a zone. Indicator species analysis is a technique that calculates an indicator value (0-100) for species (or traits) across specific groupings; then, a Monte Carlo method determines statistical significance (Dufrêne and Legendre 1997; McCune and Grace 2002). For all statistical analyses, we set the alpha value at 0.05.

Comparison Across Zones

Since their establishment, core zones have been exempt from direct human influences, whereas the buffer zone forests are still managed and harvested, albeit in a sustainable manner. Further, previous land use likely influenced the current zonation. One of the most striking differences between zones is the shape of the size-distribution curve (Figure 4). The core zone exhibits a distribution that has a relatively negative exponential shape, which is generally indicative of an uneven-aged forests (Lorimer and Krug 1983; Messina and others 1997) and has been noted in both old-growth and maturing secondary growth bottomland forests (Nixon and others 1977; Robertson and others 1978). The shape of the distribution deviates from the negative exponential shape in the smallest size-class. Lorimer and Krug (1983) suggest that such deviations may relate to canopy disturbances or deer browsing, both of which apply to these forests.

At least one of the sites experienced timber harvests immediately before establishment as a “total reserve,” and as we noted in the previous section, deer browsing has increased in the Middle Elbe forests. The core zones also possess more stems in the largest size-classes, demonstrating the greater age of parts of those forests. In contrast, the buffer zone forests demonstrate a more normal-shaped distribution, which is more indicative of a more even-aged stand (Lorimer and Krug 1983). Indeed, the forests of the buffer zone span a much smaller age-range than their core zone counterparts (Table 1).

Despite the differences in size-class distributions and disturbance regimes, we do not see any variation in canopy cover across zones (Table 6). This likely relates to two factors.

First, the sampling methodology excluded recently harvested areas that would have lowered the canopy cover for buffer zones. Second, the core zone possesses remnant poplar stands that were planted in the 1950’s for wood production, whereas equivalent stands in the buffer zone have generally been converted to other stand types (Kützner 2000; Schmidt 2002). In natural Central European floodplain forests, poplar stands represent an early stage of succession (Dister 1996; Schnitzler and others 2003).

Since the trees in the core zone stands have matured, the stands are starting to degenerate and thus possess a more open canopy.

In some respects, the remnant poplar stands replace some of the successional dynamics loss to the decreased flooding regime. As poplars are extremely shade intolerant, they do not reproduce under any canopy, and are thus replaced by more shade tolerant, later-successional species. This parallels the natural progression that usually takes place from early-successional willow and poplar communities.

Currently, the Elbe lacks large areas where such early successional stands could develop naturally due to decreased flooding, channel training structures and a lack of sand bars and islands (Gurnell and Petts 2002; Puhlmann and Jährling 2003). Thus, these stands have functioned surrogates for their natural counterparts.

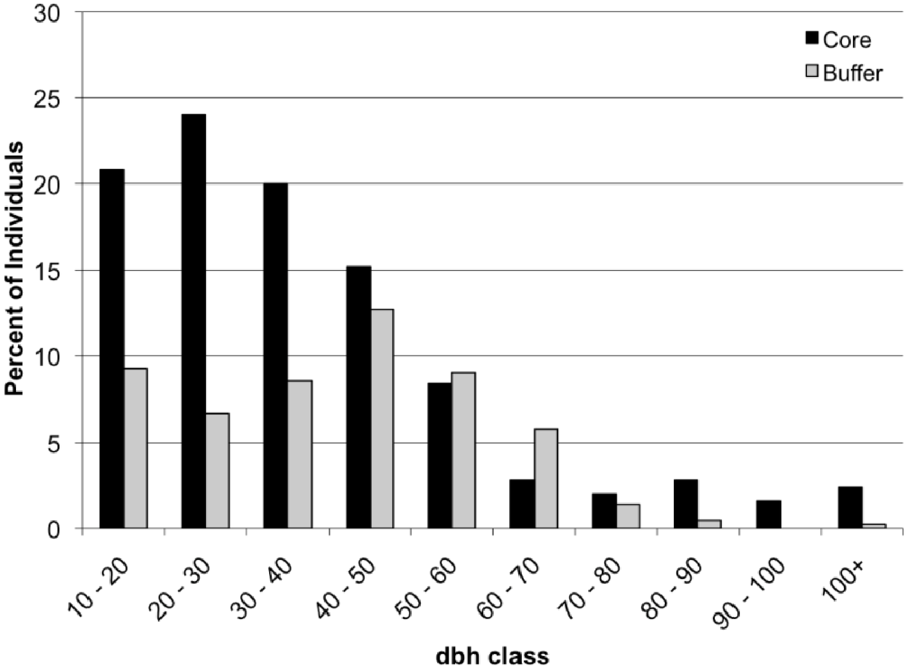

Figure 4. Size (cm) Class Distribution across Management Zones.

Table 6. Site and Forest Characteristics across Management Zones

Characteristic	Core (n=3)	Buffer (n=3)
Average distance from river (m)	308	321
Average canopy cover (%)	93	92
Average basal area per tree (m^2)	0.143	0.155
Average % of large trees (dbh > 60 cm)	11.9	14.6

Pedunculate oak is less dominant in the core zones than in the buffer (Table 7). Although the oak will remain the dominant species in the tree layer for decades to come (barring any major disease or insect outbreak), it will likely slowly decrease in importance in the core zones and eventually cease to be a major canopy species there, should no large external disturbances occur that would greatly increase light reaching the understory.

In its place, we will likely see the forests of the core zone dominated by maple and ash species, as well as with moderate levels of hornbeam, lindens, and white elms. However, the oaks will likely persist in the buffer zones, as they are commonly replanted after harvests along with a mixture of ashes and elms, and receive some protection from deer browsing (Kützner 2000; Schmidt 2002). The tree layer of the core zones also contains more of the typical natural floodplain species, such as higher levels of the European ash, wild apple, and wild pear. In contrast, the tree layer of the buffer zones contains higher levels of introduced species, such as green ash and horse chestnut, an indication of greater forestry impacts. The diversity indices and analyses do not indicate major differences between the two zones (Table 8).

The results from the understory provide stronger indications of a divergence in the trajectories of core and buffer zones (Table 9). The buffer zone has a higher understory density, reflecting the likely higher light levels of the understory, given the lower basal area and tree layer density in the buffer zones. Although neither the MRPP nor the Spearman rank correlation analysis indicates significant differences between the understories of the core and buffer zones, our results suggest that perhaps given more time, these two communities will become more different. A greater proportion of the understory in the buffer zones belongs to elm and hawthorn species, neither of which can become canopy species. Elms are one of the species typically replanted after harvests in the buffer zones, which may explain part of their larger abundance in this part of the floodplain. Higher levels of deer browsing may explain the hawthorns' abundance, as we discuss below. Of the remaining tree species, only field maple shows a moderate level of reproduction, suggesting some problems for the future of the buffer zone stands.

However, the artificial regeneration that occurs in the buffer zones should offset this trend with respect to oak and European ash. In contrast, the core zones exhibit lower levels of elms and hawthorns in the understory, although they are still present at high levels. Moderate regeneration is present for field maple, linden, and European ash. An indicator species analysis found that two species are significantly correlated with the core zones, European ash (indicator value=59, p=0.04) and linden (indicator value=50, p=0.01). Linden's presence in the core zone sites used for the study derives from pre-core zone plantings, and thus is not a result of natural ecological processes; however, its level of regeneration likely demonstrates the decrease in flooding in the core zones. European ash's indicator status for the core zone represents, however, a positive sign for those areas.

Table 7. Tree Layer Characteristics across Management Zones

Species	basal area (m2 *ha-1)		density (ha-1)		IV 200	
	Core	Buffer	Core	Buffer	Core	Buffer
Quercus robur	16.5	16.9	68.1	72.6	67.9	80.6
Acer campestre	5.2	3.9	52.7	40.2	32.7	29.0
Fraxinus excelsior	5.2	4.4	52.2	19.8	32.6	21.3
Fraxinus pennsylvanica	3.1	4.3	26.6	31.4	17.9	26.4
Populus spp.	4.0	0.0	8.3	0.0	13.5	0.0
Tilia cordata	0.7	0.0	16.4	0.0	7.9	0.0
U. laevis	1.8	0.8	8.6	7.4	7.9	5.6
Ulmus minor	0.4	0.3	16.4	19.3	7.2	9.5
A. pseudoplatanus	0.4	0.8	10.5	5.2	4.9	4.5

Malus sylvestris	0.4	0.0	2.5	0.0	2.0	0.0
Corylus avellana	0.1	0.0	4.2	0.0	1.7	0.0
Pyrus pyraster	0.2	0.0	1.5	0.0	1.1	0.0
Carpinus betulus	0.1	2.4	1.6	15.0	0.9	13.4
Aesculus hippocastanum	0.1	0.8	1.1	3.3	0.7	3.8
Crataegus spp.	0.0	0.1	1.5	3.5	0.6	1.8
Cornus sanguinea	0.0	0.0	1.1	0.0	0.4	0.0
Acer platanoides	0.0	0.0	0.0	0.9	0.0	0.5
Betula pendula	0.0	0.1	0.0	2.8	0.0	1.4
Prunus padus	0.0	0.2	0.0	2.7	0.0	1.7
Rhamnus cathartica	0.0	0.0	0.0	0.9	0.0	0.5
TOTAL	38.4	34.9	273.0	225.0	200	200

Table 8. Community Composition Indicators between Management Zones

	Tree Layer		Understory	
	Core	Buffer	Core	Buffer
Average S	10	9	10	11
Average H'	1.81	1.52	1.87	1.77
Average J	0.78	0.70	0.82	0.74
Species ranking	$p<0.05$		$p<0.05$	
MRPP	$p>0.05$		$p>0.05$	

Table 9. Understory Characteristics across Management Zones. Significant indicator species bolded

	density (ha^{-1})		frequency		IV200	
Species	Core (n=3)	Buffer (n=3)	Core	Buffer	Core	Buffer
Ulmus spp.	77	51	39%	52%	46.5	53.0
Crataegus spp.	73.8	63.1	35%	40%	43.5	54.3
Acer campestre	42.8	10.4	16%	16%	22.7	13.4
	density (ha^{-1})		frequency		IV200	
Species	Core (n=3)	Buffer (n=3)	Core	Buffer	Core	Buffer
Fraxinus excelsior	21.7	5.0	14%	6%	14.5	5.8
Corylus avellana	11.8	5.5	20%	3%	14.3	4.2
Tilia cordata	14.4	0.0	16%	0%	13.3	0.0
Fraxinus pennsylvanica	13.8	3.6	11%	11%	10.5	7.3
Cornus sanguinea	15.2	6.5	9%	14%	9.7	10.2
A. pseudoplatanus	10.5	5.8	6%	18%	6.8	11.5
Prunus spinosa	9.6	2.8	6%	5%	6.5	4.1
Sambucus nigra	0.0	10.7	10%	15%	5.1	13.1
Prunus avium	1.8	0.0	5%	0%	3.0	0.0
Carpinus betulus	2.8	6.0	2%	8%	2.2	7.0

Malus sylvestris	0.0	0.0	2%	0%	1.3	0.0
Betula pendula	0.0	2.8	0%	3%	0.0	2.8
Euonymus euonymus	0.0	1.4	0%	3%	0.0	2.0
Prunus padus	0.0	3.6	0%	15%	0.0	9.1
Rhamnus cathartica	0.0	1.4	0%	3%	0.0	2.0
TOTAL	295	179	190%	211%	200	200

As Figure 5. shows, both zones have similar levels of flood tolerant species, demonstrating that both management strategies still must cope with external impacts on the forests, such as the changes in the river's flow regime. The higher value of flood intolerants in the core zones represents the relict stands of linden in the core zone, which continue to reproduce well. Species with intermediate shade tolerance make up the majority of the understory in both zones, although performing an Indicator Species Analysis on life history traits revealed that intermediate shade tolerance (indicator value=54.1, p=0.03) was a significant indicator for the core zones. The lack of early successional species in both will continue to be a problem, as disturbances from floods remain at a decreased level. The abundance of shade tolerant species should increase in their place, as successional processes proceed. Although we expected to see higher levels of shade tolerant species in the core zones, the harvests in the core zones prior to the core zone designation, as well as the relatively open poplar stands have not yet created an environment where shade tolerant species dominate.

A surprising result is the apparent higher level of deer browsing in the buffer zones, suggested by the higher proportion of both unpalatable species and armed species in those areas. As deer often prefer younger stands where understory vegetation is denser, this finding is intriguing (Scherzinger 1996). In our previous paragraph, we suggested there are more earlier-successional stands in our core zone sampling. However, buffer zones contain a higher number of younger stands as a whole, as timber harvests continue to take place there on a regular basis.

The core zones, on the other hand, have been harvest-free for over thirty years and will remain so in the future. Further, buffer zones, by nature, often border agricultural fields, which also attract deer (Scherzinger 1996). While our finding is a relatively positive indication for the core zone forests, its implications for the buffer zones will require further monitoring and potential adjustments to deer control policies.

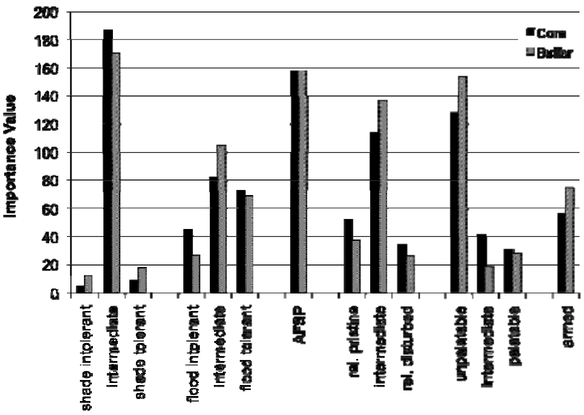

Figure 5. Life History Groups in the Understory across Management Zones.

The differential impact of deer browsing may also be one explanation for the diversity results of the buffer zone understory, which had a lower evenness and a lower overall diversity (Table 8). The buffer zone forests also had a lower importance of species with a higher conservation value. Logging impacts and impacts from higher recreational use of the forests could also contribute to decreased levels of biodiversity in the buffer zones (Dister 1996; Cole and Spildie 1998; Leung and Marion 1999; Sutherland and others 2001). However, these forests also have fewer species with a lower conservation value and approximately the same amount of ancient forest species. Thus, while there are several indications that the core zones are indeed reserves of higher quality forests, the buffer zone forests also maintain some level of ecological quality.

OTHER MANAGEMENT ACTIVITIES

As noted in the introductory section of this chapter, river-floodplains are dynamic systems in their natural state. Periodic flood and drought events drive geomorphic and ecological changes on the floodplain that support the characteristic diversity of organisms and habitats. When human interventions alter the flow regime, many natural renewal processes cease to operate. Thus, the Middle Elbe has experienced changes in flow regimes due to dams, which have reduced the importance of flood tolerance among the flora of the floodplain; a fixation of its channel, which prevents channel migration and floodplain renewal; and the construction of a system of levees, which also disrupts flooding and renewal processes on the floodplain (Puhlmann and Jährling 2003). The biosphere reserve managers have recognized these threats and have undertaken (and are undertaking) some exciting projects to help mitigate the impact of activities that fall outside of their sphere of control). We conclude this chapter with a summary of two of these projects.

EU-LIFE Project

The EU-LIFE project was initiated in the Klieken floodplain of the Middle Elbe Biosphere Reserve in 2001 to address several major problems (Puhlmann and Jährling 2003). In the area of Kurzer Wurf, a quasi backwater had been created in the 1930's, when a channel was built to bypass a meander in the Elbe. The original channel was left partially intact, but disconnected upstream from the main river flow. This had negative impacts on the structure of the river, as well as the local flora and fauna. To restore the old meander, the original connection to the main channel was restored.

Due to shipping interests and concern about impacts of this measure on low flows in the main channel, the channel level of the restored connection was set 0.5 m below the mean water level (BRFME 2002). This way during low flow periods, water does not flow through the old meander and remains in the main channel to maintain adequate depths for ships. Nonetheless, during normal and flood flows the old meander now experiences typical river flows.

An additional benefit was the creation of an area suitable for core zone designation on the island (Matzwerder) created by the restored connection. Additionally, the Klieken floodplain has a large backwater (Alte Elbe) that was suffering from severe eutrophication due to runoff from the surrounding farmland. Since no new backwaters are able to form naturally and since the rate of eutrophication had increased rapidly due to fertilizer run-off, it was decided that the lake would be restored by dredging and that a 40m buffer would be built to protect the restored lake (Puhlmann and Jährling 2003). The initial results from these projects have been very positive.

WWF Great Project

One of the major issues of the Elbe floodplain is the large area that no longer floods due to levee construction. In areas where local flood protection is no longer necessary for the human settlements, leaving the levees intact prevents vital floodwaters from reaching the natural ecological systems.

Further, on a larger scale, the floodwater buffering capacity of the floodplain remains unnecessarily reduced, which could otherwise protect downstream settlements from floods. Perhaps in no area is this more noticeable than the levee bisecting the Lödderitzer Forest—the original core of the biosphere reserve and one of the largest stands of floodplain forests along the Middle Elbe. The idea of shifting this levee back further in the floodplain dates back to at least 1994 (Jährling 1994).

As of 2009, this project will become a reality. Working together with WWF, the Biosphere Reserve has won local support and approval to set back 7 km of levees running through this area (WWF Deutschland 2009). When completed, 600 ha of floodplain will be reconnected to the flow regime of the Elbe, which will help restore the forests, meadows, and backwaters that are located there.

Further, this area will also represent an additional 600 ha of floodwater storage space when the next major flood occurs. This levee project is one part of WWF's "great project" (Grossprojekt) on the Elbe that also includes the restoration and reforestation of former military lands and the creation of 9000 ha of interconnected, active floodplains.

CONCLUSION

The floodplain forests of the Middle Elbe have experienced great changes over the past 50 years, relating to decreased flooding levels, forestry management practices, increased levels of herbivory, and introduced disease and species. Historical forest composition and structure was a product of intensive forest management and flood regime. Forest management exaggerated the presence of the pedunculate oak, which shared dominance with two other floodplain species, common ash and the small-leaved elm (Figure 6). In the last 50 years, the decrease in flooding, changes in forest management policies (including the establishment of a biosphere reserve), and Dutch elm disease have altered the floodplain communities. Pedunculate oak has benefited greatly from the changes and become the dominant species in both management zones. The current tree layer differs across zones, primarily in the much greater presence of the exotic green ash in the buffer zone. However, we anticipate that, with time, differences should become more pronounced. Both communities will experience increasing populations of less-flood tolerant species, such as sycamore maple. The core zones may see the oak eventually disappear as a dominant species, whereas forestry practices should sustain its presence in the buffer zones. In general, different management practices in the core and buffer zones will maintain a greater diversity of forest types and successional stages across the region, which will be important for preserving the wide range of flora and fauna that characterize the region and the biosphere reserve (Dornbusch and Dornbusch 1991). Management staff will continue to face challenges from herbivory and the spread of exotic species. However, the decrease in flooding regime remains the dominant factor driving changes in these forests, and mitigating its effect will remain much more difficult.

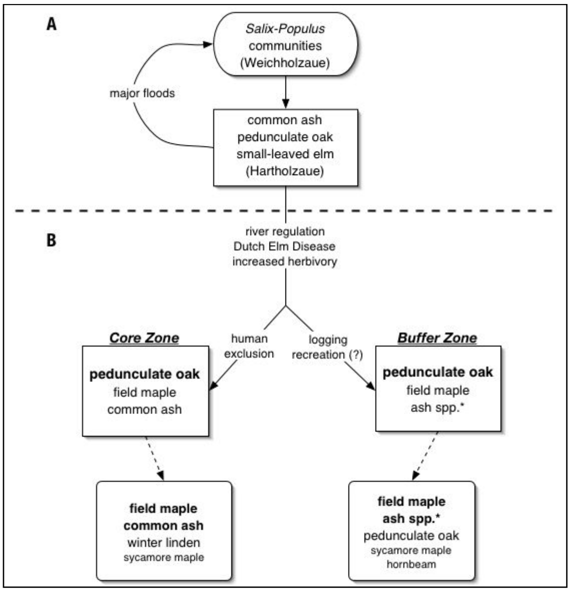

Figure 6. Successional dynamics of floodplain forests along the Middle Elbe River. Part A derives from Passarge (1953) and describes the former successional process. Part B presents the current (solid line) and possible future (dotted line) successional dynamics for forests in the core and buffer zones of the Biosphere Reserve. Font size and shading of species names reflect their relative dominance in the community. *indicates both the dominance of both common and green ashes.

APPENDIX 1.

Species	Shade Tolerance	Flood Tolerance	Ancient Forest Species	Conservation Value	Roe Deer Palatability	Herbivore Defenses
Acer campestre	intermediate	intermediate	yes	high	low	no
Acer platanoides	high	low	no	low	low	no
Acer pseudoplantanus	high	low	no	low	low	no
Aesculus hippocastanum	high	intermediate	no	low	low	yes
Betula pendula	low	low	no	intermediate	intermediate	no
Carpinus betulus	high	low	no	intermediate	high	no
Prunus avium	intermediate	intermediate	no	high	low	yes
Cornus sanguinea	intermediate	intermediate	yes	high	high	no
Corylus avellana	intermediate	low	yes	high	high	no
Crataegus spp	intermediate	intermediate	yes	intermediate	low	yes
Euonymus	high	low	yes	high	high	yes

europaea						
Fraxinus excelsior	intermediate	high	no	low	intermediate	no
Fraxinus pennsylvanica	intermediate	high	no	low	intermediate	no
Malus sylvestris	low	intermediate	yes	high	high	yes
Prunus padus	intermediate	intermediate	no	high	intermediate	no
Populus spp.	low	high	no	low	low	no
Prunus spinosa	intermediate	intermediate	no	high	low	yes
Pyrus pyraster	low	intermediate	yes	high	high	yes
Quercus robur	low	high	no	high	high	no
Rhamnus cathartica	high	intermediate	yes	high	low	yes
Sambucus nigra	low	low	no	low	high	no
Tilia cordata	intermediate	low	yes	intermediate	intermediate	no
Ulmus laevis	high	high	yes	intermediate	low	no
Ulmus minor	intermediate	high	yes	intermediate	low	no

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Chapter 4

FROM "ALL STAKEHOLDERS" TO BALANCED PROCESS IN PARTICIPATORY PLANNING: A PRACTICAL, STEPWISE ETHICS

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ABSTRACT

In river management as in most other fields of environmental planning, "all stakeholders" are supposed to somehow participate. But who will actually sit in, who will actually have a voice? This question is of great consequence in practice, but underdeveloped in literature.

Taking most of its examples from the field of river management, the present paper aims to shed some light on the dark road that leads from "all stakeholders" to a balanced representation of stakes in participatory, collaborative planning.

Our method is to develop the initially undifferentiated cloud of stakeholders through five steps: (1) from all stakeholders to significant stakeholders, (2) framing reflection, (3) representation of significant stakeholders, (4) setting a balanced scene, and (5) maintaining a balanced process.

We then attach a number of categories, examples, reflections and sometimes a few rules to each of these steps. This system serves as an informal ethics to improve the quality and justifiability of who is to represent what in participatory planning.

Keywords: stakeholder analysis, participation, multi-stakeholder platforms, collaborative planning, joint planning, community-based planning, river management, river policy.

INTRODUCTION

From clean air to fish stocks and rural landscapes, environmental goods contain values on many levels and for many people. On the local level, these goods are used and cherished by inhabitants, farmers, recreationers and many others. And on the higher, supra-local scale, environments hold values for national biodiversity, economy and culture. River systems and river landscapes are a prime example of this multi-value and multi-stakeholder phenomenon, playing as they do important roles in production systems, cultural identities, species and ecosystem diversity, water balances, recreation, transport and so on. In this chapter, we will take our examples from the river management field but the issues we will discuss are applicable to basically all types of environmental and landscape planning.

In the last decennia, Western democracies have become more 'deliberative'. The role of the state as the paternalistic and rational representative of 'everybody' and the common good is not taken for granted anymore, giving rise to all sorts of forms of 'co-management', 'multi-stakeholder planning', 'participatory decision-making' and so on. New challenges accompany this shift. River management authorities and professionals, for instance, long used to the idea of rational embodiment of *all* values in their comprehensive planning methods and culture, have to unlearn many explicit and tacit routines, and find new ways to relate to communities and citizens.

These new ways are not only technical. Many challenges of participatory planning are ultimately *ethical*. For instance, if the river authority still sees it as its duty to embody all values in its river policy and project designs, does giving a voice to local communities and user groups not automatically amount to double-counting, in fact giving a double weight to local interests? Or, if the authorities would withdraw to only a role of process facilitators, who will represent the supra-local 'system rationality' of rivers and the supra-local values that rivers have? Or, if the authorities would take it upon themselves to represent these supra-local values, how can the authorities act as facilitator and as a representative of values at the same time? Or, if the river authorities would withdraw to only that role of supra-local value representation, would that not inscribe automatic conflict in any of its interactions with local communities? And apart from all that, how to know and prevent that the local voice may dominated by a only few articulate interests? It is against this background that in the overview of Decoster (2008), an "acute sense of ethics" is listed among the key qualities of the planning professional.

In the present paper, we will focus on one problem that holds a key position and that is at the same time scarcely discussed in the literature. This is a problem of balancing, namely of how to prevent that some actors get a disproportionate say in the process, at the cost of others. Therefore, we will seek to gain conceptual ground and tease out some of the practical ethics of *who is a stakeholder and who will participate in the planning process*.

On that basis, we present some suggestions for the balancing act that is often one of the most decisive, even if least visible, roles of the planning professional in a participatory context. For ease of reading and memory, these suggestions will be ordered in a stepwise manner. This is not to say that all steps should be followed in a rigid manner. There is a logic to begin with step 1, of course, but already step 2 may result in a revisit of step 1. And step 5, "maintaining a balanced process" is of course not literary a step but more like a continuous activity.

Underlying the ethical challenge of the balancing act are a number of conceptual issues of what in fact are “stakeholders” and “participants”. Parti-cipation, from the Latin, is ‘taking part’. Much literature refers to ‘stakeholders’ while in fact they mean participants. Clearly, however, not all stakeholders can or will take part in a public planning process, and other actors may participate even without holding a significant stake. What does it then mean when it is characteristically said that “the” or “all’’ stakeholders should or did “participate” in a planning process? All too often, it means that choices of inclusion and exclusion have been obfuscated, since all stakeholders are usually too many to participate. For this reason, we will pay much attention here to unpack the distinction between stakeholders and participants. Having gained this conceptual ground, elements of a practical ethics of balanced representation will become visible with greater clarity than in the often opaque routines of planning practice.

We do acknowledge that social and political life often benefits from conceptual opacity, e.g. because fungible words may smoothen relationships, buy time, enhance creativity and obscure winners and losers, as opposed to conceptual transparency that may promote conflict and innovation (Harrison and Hoberg 1994; Bal and Halffman 1998). The planning professional, however, should be able to make a conscious choice between the one way or the other, and conceptual clarity is prerequisite for that capacity.

The structure of the chapter follows the stepwise approach. The next section starts out with a definition of the key terms and then explores some ethical ramifications of mainstream ideas on who is stakeholder and who should be invited to take part in the planning process. The subsequent section focuses on the first step along the way that leads to balanced representation, namely the identification of significant stakeholders. We then illustrate some of the subtle language games played in public planning, which is one aspect of the significant stakeholders issue and leads to the ‘framing reflection’ step 2 in our practical ethics. The next section moves on to step 3, focusing on who may be seen as representing the stakes selected as significant, both separately and as a (balanced?) group. Subsequently we discuss the step towards actual participants in the planning process. Finally we focus on some of the issues involved in keeping up the representation balance in the heat of the ongoing participation process.

STAKEHOLDERS, PARTICIPANTS, ETHICS

Ultimately, everybody is a stakeholder in everything. Just like the ‘five handshakes’ game linking everyone to everyone else on earth, everyone is linked to different degrees to any particular policy topic. In environmental policies, the physical effects of decisions can be surprisingly wide, most clearly illustrated by the interdependencies around CO_2 emissions. Financial effects of government decisions ultimately regard all taxpayers. ‘Stakeholder’ thus is a catch-all concept. In the past decades, various pieces of legislation have started from a concept of participation in which all stakeholders can be participants. The 1998 South African water law, for example, states that all stakeholders may participate, leading to some 50 million potential participants in water policymaking. Similarly, the stakeholder identification clause of in article 14 of the European Union’s Water Framework Directive (WFD) prescribes the "active involvement" of all interested parties, thus inviting all 360 million or so Europeans to any planning table in the Union.

The waters at the confluence of "stakeholders" and "participants" are muddy. As said, this may sometimes allow good political fishing but it obscures the fact that between these two categories lie pertinent practical and ethical issues of representation, inclusion and exclusion. *De facto*, organizers of participatory processes do not want everyone to show up and sit in. Even the famed democracies of the ancient Greek city states did not allow everyone to participate. Men without property, women and slaves were excluded from the *agora*. Any participatory process not only means an opening up of the range of actors, but also a closing (Stirling 2008), if only to keep numbers manageable. Balanced participation requires that some rather than all should be involved, representing others who have other business. Since the term 'stakeholder' does not define anybody in or out unless a qualifier is added (primary/secondary stakeholders, salient/latent stakeholders etc.), we need additional concepts to denote who is or should be in or out.

Let us start out with the current definitions of stakeholders and participants in connection with public policy problems. For the purposes of the present article, we will broadly follow Hemmati (2000), Mikalsen and Jentoft (2001), Margerum (2002), the World Bank (Borrini-Feyerabend et al. 2004) and most other mainstream literature, defining that

> "stakeholders are any group or individual who can affect or be affected by the problems at hand or the possible solutions".[1]

Note the terms of 'affect' and 'be affected'. Stakeholders are found on both sides of problems and solutions. On the side causally downstream of the problems and solutions, they are the actual and potential recipients, the victims, the beneficiaries. On the upstream side, they are the actors that actually or potentially cause the problems or solutions, e.g. directly cause the problem, indirectly cause the problem, facilitate the solution or may block it. Allan (2005) uses the term 'problemshed' to denote the (spatially and temporally) linked actors and factors that affect or are affected by a problem or policy.

The only drawback of this stakeholder definition is that it suffers from misplaced concreteness at one point. For instance, if we say that sustainability is an important policy goal we do define a stake. But where is the affected group? Are "future generations" a group? They do not exist yet, so how could they hold a stake and participate in planning? Likewise with biodiversity. If its conservation is stated as a policy objective, we acknowledge that there is a morally considerable stake out there in nature, but who is the stakeholder? The great risk connected to these questions is that stakes that cannot readily be seen as held by some concrete, addressable actor group are not considered to enter the stakeholder list or are dropped at a very early stage. Getting them safely on the list and keeping them there long enough to take part in Step 1 of our practical ethics requires a first focus on relatively abstract *stakes* rather than too concrete stakeholders. Supporting this idea, our formal definition of stakeholders is:

stakeholders are any morally considerable entity representing a stake (interest) that can affect or be affected by the problems at hand or the possible solutions.

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[1] Charmingly, this definition is the opposite of the original meaning, in which the stakeholder was the uninterested party holding the stakes for parties engaged in a bet or a duel (Mitchell 2001).

Moreover, the overview list compiled to prepare a planning process may well be a mix of stakes and stakeholders both. It is better to start out with a somewhat incoherent inventory of stakes (recreation, mining, drinking water, gender, biodiversity, farming, health, children etc.) and stakeholders than with a too limited list of only stakeholders. Our stepwise procedure in the coming sections takes care of a sensible route from stakes to participants.

We take the definition of participants from its literal meaning and define that in public planning,

participants are any group or individual who actually sit in (around the table or behind the curtain) to witness and discuss the diagnosis, analysis and explanation of a public problem or the designs of its possible solutions.

In river management, problems are typically written in terms of water quality, flood risk, drought vulnerability, landscape quality loss, degradation of ecosystem services or biodiversity loss. Possible solutions are typically the well-known physical options of pollution abatement, river regulation and river renaturalization, added to which are all sorts of social and institutional changes (i.e. changes in rules or organizations) needed to realize the physical options in practice (De Groot 1998).

Some literature does make a difference between stakeholders and participants but then fails in thinking through the ethical ramifications of the distinction. One illustration is the otherwise excellent overview by Mikalsen and Jentoft (2001) of mainstream stakeholder theory developed in business management. Who out of the stakeholder category will come to count as actual participants? Their answer is that this depends on "stakeholder salience". Salience is scored according to three criteria:

- legitimacy (having an interest, *i.e.* a legal or moral claim)
- power (*i.e.* the degree to which the actor can affect success of the plan; 'primary' stakeholders are essential to success, 'secondary' stakeholders are not)
- urgency (not defined by Mikalsen and Jentoft).

Mikalsen and Jentoft then propose to score the stakeholders on these three criteria and define:

- "definitive stakeholders" as those with a plus on all criteria,
- "expectant stakeholders" as those with two plusses,
- "latent stakeholders" as those with only one,
- and the stakeholders highest on the list become participants.

There are problems with this approach. If we differentiate between primary and secondary stakeholders this way, victims of a decision who have little power can never be defined a primary stakeholder and participate, regardless of the degree to which they are affected. The whole of the coastal population around the Gulf of Mexico would this way be excluded from participating in deliberations on the BP oil disaster. To take an example from river management, a loud-mouthed local political boss whose cellar might become somewhat moist due to a possible rise of the groundwater level would have all plusses and be invited to participate, while a poor household whose whole house may be demolished due to a dike relocation would never be included. This may be already dubious for business management; it is morally fatal for public policies.

A second illustration of ethical ramifications is offered in Brugha and Varvasovszky's (2000) review of stakeholder analysis. After the identification of stakeholders, the analysis starts with data collection (usually by interviews) and then proceeds to study

- the stakeholders' involvement in the issue
- the stakeholders' degree of influence
- the stakeholders' interest in the issue
- the impact of the issue on the stakeholder and
- the position of the stakeholder in the issue (*i.e.* whether they are supportive, non-mobilized or opposed to possible solutions).

In Brugha and Varvasovszky's approach, the information on the position of the actor is used to find the "optimum fit" between the policy-proposing organization and the stakeholders. This optimum fit is that marginal actors should be monitored, supportive stakeholders should be involved and opposing stakeholders should be defended against. Brugha and Varvasovszky's work stands in a perspective of effective manipulations to push a desired policy change through. Whatever this may be, it is not an ethics of public participation.

All this highlights the importance of stakeholder analysis and especially of stakeholder selection, which defines the in-group of participation. This is the subject of the coming sections.

STEP 1: SIGNIFICANT STAKES AND STAKEHOLDERS

How to arrive at a good first list of stakes and stakeholders as potential participants in the planning process? We will here call *significant stakes and stakeholders* all those that are affected or affecting the problem or solutions above a certain threshold. Significant stakes and stakeholders form the long-list out of which planning participants will be short-listed later. Often, stakes and stakeholders have a simple, one-to-one connection, such as between the stake of 'recreation' and the stakeholder group 'recreationers'. In such a case we can put the stakeholders on the list, with the advantage of concreteness. Recreationers can be talked with, for instance. But if stakeholders are not clearly visible yet (e.g. in stakes such as biodioversity, sustainability, cost efficiency etc.), we should retain the stakes and worry about their representation only later (in step 3).

Roughly, identification of significant stakes and stakeholders may be carried out on three levels of precision. The first level is to simply follow the experts' own insights in the problem situation and the possible solutions. The second level consists of interviews with informants and stakeholders, in which respondents may point at others who might also be affecting or affected (Brugha and Varvasovszky 2000).

The third level of precision in stakes and stakeholder identification uses more or less formal frameworks. These frameworks often emphasize either the 'affected' (victims) or the 'affecting' (causers) side of the stakeholders category. A comprehensive classification of all ecosystem services of a landscape, for instance (e.g. De Groot 1992: 229) can easily be connected to affected stakeholders. Kessler's (2003) regional planning framework SEAN-ERA starts out from such a classification. By contrast, Imperial (1999), in a paper on ecosystem-based planning based on the work of Ostrom (the Institutional Analysis and Development Framework, IADF), focuses on the affecting side of the stakeholders group, involving all those who make decisions affecting the ecosystem. His framework does include equity as an evaluation criterion, however, as if to make good for neglecting the affected parties. De Groot’s (1998) framework called Problem-in-Context covers both the affected and affecting sides of environmental problems. The affected stakeholders are represented through their connection with impact variables such as health, incomes, well-being, biodiversity and sustainability, while the affecting stakeholders are identified by a causal sub-framework called Action-in-Context. This includes an ‘actors field’ analysis in which actors that directly cause environmental change (‘primary’ actors such as farmers, tourists, transporters) are connected to ‘secondary’ actors, defined as those influencing the actions of the primary actors, by way of exerting power over the latters’ options and/or motivations for action. Then follow the tertiary actors, and so on. This structure of affecting stakeholders may run all the way from, say, floodplain farmers to absentee landowners, banks, urban NGOs, government agencies and so on. All these actors affect the problem; hence all may be significant stakeholders. The advantage of formal frameworks such as SEAN-ERA, IADF or Action-in-Context is that they help avoid the trap of privileging known, articulate and well-organized groups and ignoring less vocal, non-human or unborn stakeholders. The relatively abstract nature of frameworks may also help avoid slipping from abstract stakes to concrete stakeholders too quickly. If for instance a policy proposal entails an incentive for farmers to switch from food production to a non-food crop, food production is an affected stake. The connected stakeholders, however, are not the farmers but rather the general public, which (if significant) might be represented in this case by the ministry of agriculture, e.g. if national food self-sufficiency is a policy goal.

The real world, as always, tends to resist the neat objectivity of methods and frameworks. To mention one example, stakeholder analysis tends to work with identities that appear to reasonably label a group, e.g. farmers or recreationers. People can be categorized along many dimensions however, e.g. economic, occupational, demographic, geographic or cultural. Which dimension is most appropriate for the problem at hand? Moreover, the stakeholder identities themselves are not static. In developing countries, for example, fishermen may also be part-time taxi drivers, and industrial laborers next month. Also the project itself may impact on actor identities, e.g. when dam oustees become street peddlers. Finally, problems are dynamic, and also new solutions (and, alas, new problems) may crop up in rapid succession. With that, the whole array of affected and affecting stakeholders shifts. We will pay more attention to this in Step 5.

Step 1 in our practical ethics of participant selection is rounded off by applying a threshold level to select significant stakes and stakeholders from among the often wide range of all stakes and stakeholders. Such a threshold will usually have been applied already in an implicit way during the identification process. For instance, any river management project is bound to cost public money and with that, all taxpayers are affected stakeholders, formally. It is however quite possible that they have never been taken up in the first inventory at all, for two reasons. First, we may assume that the river authority, mandated as it is through a democratic process and bound to its budget, will take care of the need to search for cost-effective solutions. Second, the level of being affected is too low to be of ethical relevance; for 2 dollar per taxpayer, for instance, you don’t call them all stakeholders. In such a case, the abstract stake of cost efficiency is connected to only the river authority as stakeholder. This example shows that any justifiable selection of significant stakeholders involves good reasons as well as good inclusion threshold levels. We cannot specify here what good reasons and good levels exactly are. We assume that once planners are aware of the need of justifiability at these points, great progress is already achieved compared to the usually implicit selection. In other words, we assume that once a concept such as exclusion reasons is made explicit, planners and other actors will more or less automatically grasp that these reasons should be coherent and applied even-handedly. As for the threshold levels of inclusion, it may be noted that high levels result in few significant stakeholders and low levels in many. Inclusion threshold levels do not need to be the same on both sides of the problem. A high threshold on the affected stakeholders side combined with a low level on the affecting stakeholders side will tend to result in a planning process with much voice to the powerful stakeholders that can make or break solutions. This can be justified if the interests of affected stakeholders are well protected (e.g. by law) and implementation of solutions is so difficult that a concerted effort of affecting stakeholders is needed. The reverse can also be true, of course. In other words, justifiable levels of inclusion are levels in context.

STEP 2: THE FRAMING REFLECTION

Step 1 of our practical ethics has resulted in a 'long list' of significant stakes and stakeholders out of which participants will later be selected. Step 2 amounts to taking a self-critical look at this result. If we have a certain situation with flood risk, farmers, biodiversity, floodplains and communities, shall we call this a river management problem? Or a spatial planning problem? Or also a landscape quality problem? And possibly a poverty problem or a gender problem too? Such decisions bring whole groups of stakeholders in or out. Moreyra and Warner (2008) have called attention to the deep impacts of what problems and projects are framed to be. Problems and solutions are not simply givens out there in the world. They are to a large extent constructed by the analyst or by actors. Any framing of a problem lifts specific physical, social and normative elements out of the real world and puts them in a box, a narrative of its own that defines certain actors, problems, powers and resources in and defines others out. As Stirling (2008) puts it, "framing thus raises important queries both for analytic and participatory appraisal. (..). It reveals the enormous latitude for inadvertent, tacit (or deliberate, covert) influence of power." Framing can be a deliberate political act, making or breaking linkages between dissimilar policy areas, and bringing whole new sets of affected or affecting stakeholders in. At Lent, a village facing the city of Nijmegen situated on a hydraulic bottleneck in the river Waal (Netherlands), a dike relocation was proposed by the river authority, which would require the demolition of several houses and therefore generated fierce citizen protest. In the course of the negotiations with Nijmegen city, a new bridge over the river Waal was made contingent on the dike relocation. This framing of solutions made it very inconvenient for the river authority to accept alternative options tabled by citizen groups, since many powerful actors in and around Nijmegen were connected to the bridge and were now all 'framed in' as stakeholders.

The events at Lent are an example of causal framing of solutions, in which solution elements are made dependent on each other. Causal framing can also concern the problem itself, declaring what connects to what in its causes or its impacts. Framing may also address the spatial and temporal characteristics of problems and solutions, e.g. the boundary between local and external worlds, or the scenarios that a solution should be a solution for. In Dutch river policies, for instance, many proposals have been rejected off-hand because they were not "robust", meaning they might have to be redesigned at some future point in time. This point in time is defined as the moment that risk models might begin to predict that a mythical flood of 18,000 cubic meters per second in the Waal could conceivably hit the nation. At present, risk models do not predict such a risk at all; the 18,000 cubic meters are typically framing, namely a (convenient) belief presented as objective truth. Being able to do such a thing is 'framing power'.

One type of framing is of special relevance to river management. It addresses the urgency and certainty of facts and values, and is depicted in Figure 1. The figure is a quadrant visualizing that values (goals, interests) at stake in the problem may be contested or uncontested, and the facts on which the problems and solutions are based, e.g. the river's expected peak flow, can be certain or uncertain. Each combination defines a frame, one example being the 'politicization frame' that occurs when facts are treated as certain but values are contested. Each frame is characterized by its own natural type and level of stakeholder participation. For instance in times of crisis, participation is limited for the simple reason that participation takes time. Note that Figure 1 may be read in both causal directions. If we reason from the axes to the centre, the figure specifies the frame that follows from the circumstances. If facts and values are uncontested, we arrive in the crisis frame. Debated values and uncertain facts call for a deliberative approach, in which many stakeholders are marked as significant and invited to participate. This direction of reading the figure is also followed, for instance, by Hisschemöller and Hoppe (1998) who call this debated/uncertain type of problems 'wicked problems' and argue that these are very suitable for deliberative learning processes. Likewise according to Bulkeley and Mol (2003), participation should help bridge the gap between scientifically defined facts and values of environmental problems and the facts and values of experienced, place-based knowledge. Funtowicz and Ravetz (1993) commend 'post-normal science' including lay and expert participants.

		VALUES	
		Not debated	Debated
FACTS	Certain	Crisis frame Imposition; no time for debate, security professionals act.	Politicization frame Struggle over values, activist citizens and lobby groups dominate.
	Uncertain	Managerial frame (risk management) Controlled decision-making by experts, lay people may be consulted.	Deliberative frame Experts and lay people learn together.

Figure 1. The participation framing matrix. Levels of contestation on facts and values defines a frame. And the reverse: who-ever has the power to define a frame therewith defines what can be contested. From the frames follow the degrees of stakeholder involvement. Based on Warner (2008).

The relationships in the Figure work the other way around too, however. In order to get a decision out fast, planners may seek to frame a problem situation as a crisis situation, so that extensive participation is naturally unnecessary or even undesirable. The framing of a problem as a security (crisis) issue is a surefire way to eliminate participation. It suggests agreement on values and facts – the house is on fire now, bring in the firemen, we'll talk about collateral damage later. Wolsink (2003) has revealed the persistent efforts in the Netherlands to introduce "special" or "emergency" legislation to exclude citizens and alternatives from decisions that are given national status because of security or other overriding concerns. Such ‘securitization’ (Warner 2008) constructs a problem such that there is no need to involve stakeholders as participants. Water safety issues offer many examples but so do the war on terrorism, the war on drugs and many others.

If it is impossible to push a crisis frame through, the next option is to declare that we all agree on the values (goals) and only the facts are uncertain. This ‘managerial frame’ puts the problem in the hands of experts, who become privileged participants. If value differences are framed to be relevant in the problem situation, the ‘politicization’ and ‘deliberation’ frames dominate. A political problem reflects a divergence of values without facts being greatly contested. Political issues may be fundamentally un-decidable and go down to the wire relying on the voting rules of representative democracy – which is why those who benefit from the status quo are unlikely to politicize an issue. Douglas and Wildawsky (1982) have shown how ‘blame frames’ work to politicize risk issues, including flood risk issues. Identifying a problem often implicates attributing blame: identifying a culprit who has occasioned the trouble. From a participation perspective, there is something to be said for non-attribution of blame, even though blame may be obvious. De-politicization opens up the deliberation frame and may be the only viable way of getting the otherwise accused party to the planning table. Finally, those who are interested in a deliberative process rather than a quick decision or a politicized blaming game, do wise to frame a problem as ‘wicked’, with both facts and values under debate. This is precisely what participation facilitators do when they woo their hoped-for participants. If this frame is not true enough, however, participants are likely to drop out later, when nothing much appears to be at stake, really.

What is the relevance of this relatively theoretical issue for our practical roadmap to travel from “all stakeholders” to actual participants? The relevance is that in most cases of public planning, the agency that performs the identification of significant stakeholders is usually at the same time an agency with much framing power. The river authority is an example. This brings a duty of a broader, ethical reflection on how the problem has been framed. What people may have been excluded due to our view of what constitutes the problem and the feasible range of solutions? Are we so eager to get things done that we have framed the problem in a crisis frame? Do we present uncertain facts as if they are scientifically undisputable in order to keep the planning in a managerial frame and retain our power to deal with the problem? Are we so afraid of the irrationalities of politics that we have kept value discrepancies under the carpet? Or are we so eager to show a modern, deliberative frame that we present the simplest issues as in need of extensive debate? In other words, are we able to really justify our reasons and inclusion threshold levels of who is a significant stakeholder?

This reflection constitutes Step 2 on the road to a balanced representation of stakeholders. It may or may not result in a revisit of step 1. In any case, one result is a much better story to explain the selection of significant stakeholders to the outside world.

Step 3: Stakes and Stakeholders Representation

Future generations and dolphins have no telephone. National authorities may be too busy to sit at every local planning table. Women or the illiterate may never show up. Downstream stakeholders may be too far away. Reasons such as these often work for such groups to *de facto* fall off the list of significant stakeholders, even though the formal (and morally proper) definition of stakeholders does not exclude actors who just happen to be hard to hear. We will discuss four sources of morally problematic exclusion here, namely distance in space, distance in time, abstract interest and socio-economic factors.

Moreyra and Warner (2007) and Wester *et al.* (2007) have shown how the geographic delineation of problems and solutions can exclude key stakeholders in practice. In river management this may concern downstream affected communities, upstream affecting communities and the representatives of wider, basin-scale rationality. Current thinking in integrated water management seeks to overcome these problems by promoting the river basin (watershed) as the best level of decision-making. Administrative boundaries are hard to ignore however, and on a more principled note, also river basins have downstream communities and upstream affecting actors, e.g. through the hydrological cycle that spans the oceans and economic dependencies. The harbors of Hamburg and Antwerp are certainly significant stakeholders for large-scale planning in the Rotterdam harbor, for instance (which is not to say that they should also be shortlisted as participants, of course). Moreover, should local river planning be discouraged for the sole reason that it is local? It would rather seem wise to have methods to represent significant non-local stakeholders in the local process.

Not only geographic distance but also the shadow of the past (cultural heritage) and the shadow of the future hang over present deliberations. Should past and future actors be excluded because they are mute? Should abstract interests such as keeping options open for future generations be ignored because they are difficult to represent? Finally, many people and policies also recognize the intrinsic values of nature. How should these be represented? And how about spiritual values (e.g. holy sites, water for baptism, the river as spirit) that do not appear to be owned by any stakeholder group? Here again, rather than quietly dropping these interests from the stakeholders list, the problem of stakeholder *representation* should be tackled.

The ideology of deliberation assumes that citizens can actively raise, defend or withdraw their stakes. But many in society cannot. A critical school of participation scholars argues that due to the political economy, the poorest have very few degrees of freedom, so that they continue to end up in the most vulnerable location, e.g. geographically (next to the city dump or in the lowest part of the floodplain). Deliberative processes tend to favor the well-educated and well-paid, and should be actively protected against their intrinsic drift to exclude the socio-economically weak segments of society. Liberal-reformist authors such as Edmunds and Wollenberg (2002) have taken on board the lack of a level playing field, the need to represent non-organized stakeholders and the need to open up space for unheard voices. In Sweden, for example, marginalized groups are actively targeted to be involved in health care decisions that affect them (Tritter and McCallum 2005). In step 1 of the process from stakeholders to participants, the formal definition of stakeholders should be rigidly adhered to. Then in step 3, the question that takes priority is: *how may the significant stakes and stakeholders identified in step 1 be represented in the envisaged participatory process?* Thus in step 3, we travel from a list of significant stakes and stakeholders to a list of persons and organizations that will be invited to the planning table. We start out with an overview of types of representation in deliberative contexts, and some of the risks involved in each.

Representation by Government Organizations (GOs)

GOs are often stakeholders directly, mainly as affecting ones, if they hold some power over the issue in question. GOs may also serve to represent general public or system-level interests on the 'affected' side, e.g. standing in for the general taxpayer or consumer, or representing basin-wide river management interests. In GOs, the 'affected' and 'affecting' roles are often combined (because representing the affected common good is legitimating public power). A well-known risk here is that often, only the affected side of the GO (represented by some nice, participatory person) will participate in a planning process, but that later in the planning process, the necessary funding and permits turn out not to be forthcoming, even if these are to be issued by the same GO. Early involvement of the other, affecting side ('power side') of the GO, e.g. represented by some higher-up bureaucrat, may prevent this. Another solution may be to represent the GO not only in nice persons but also on formal paper (see below).

Representation by Interest Organizations

Constituents may often establish organizations for purposes of representation of one or more of their interests. In developing countries these are often called POs (people's organizations), e.g. a farmers PO or village PO. Labor unions, farmer unions, chambers of commerce, local interest groups and many others are of the same nature. Other interest organizations work specifically for the provision of some collective good, e.g. drinking water, recreation or river transportation. For all those, the distinction with NGOs (below) is important because of the constituent control (e.g. one vote for each member) or government control that limits risks of elite capture. A major risk inherent in the participation of interest organizations is rigid defense of the narrowly defined interest that is their *raison d'être*. POs are often established for economic issues only. Many others are typically the sectorial agencies of drinking water, forestry, transport and so on. All of them have to take care not too deviate too far from these primary objectives. The issue here is often what in fact is and should be represented. Do the organizations defend the daily *means* by which they achieve their objectives (zoning regulations, water purification, protected areas, roads construction etc.), or the *objectives* themselves (safe drinking water, biodiversity, urban connectivity etc.)? Organizations tend to stick rigidly to their means, while the art of participation and successful conflict resolution is to focus organizations (and people) on their objectives rather than their positions or, in the words of Mitchell (2001) on what they really need.

Representation by Non-Governmental Organizations (NGOs)

Representation of stakes and stakeholders by NGOs is a widespread phenomenon, especially for voiceless stakeholders (nature, children, future generations etc.) and broadly subscribed interests that are too narrow to base a PO on (cultural interests, recreational interests etc.). NGOs are typically free-floating elite organizations that 'land on' a certain issue, without significant constituent control. This voluntary enthusiasm has practical advantages for participation, but elite capture of the issue is ever a risk. Extreme examples are found where capture is easy (e.g. due to very low literacy and power levels of the real stakeholders) as well as attractive (e.g. because of legal rights of the stakeholders). A well-known extreme case in point is when indigenous people are 'represented' by a local political boss under NGO cover, who in fact mainly represents his own wallet and power position. On the other extreme are the broad public-based NGOs representing 'unattractive' stakeholders such as WWF standing in for the interest of nature. Even in such cases, surprises are possible, e.g. when in the Maaswerken project (Netherlands) an environmental NGO clashed with local protesters over what in fact 'nature' meant: new wilderness or nature in the traditional, arcadian landscape (Warner 2008).

Representation by Informal Leaders

Communities often have local heroes who are popular participants. On the other hand, many social scientists have warned against the naïve assumption that communities exist at all (Agrawal and Gibson 1999). A town may have a business community, an ethnic community, a football community, a neighborhood community a church community and so on, but is the town itself a community? And these sub-communities, do they in fact function as communities with respect to the problem at hand, e.g. town planning or river management? Communities often exist around specific functions such as football, business or religion, and the implicit mandates of their leaders often apply to these functions only. Moreover, local leaders often wear multiple hats, giving everybody a hard time to get a clear picture of what in fact they represent.[2] One remedy against this risk, of course, is to be fully aware of it, find out what the informal leaders in fact lead, and be very selective in the final decision to invite the leader to participate. A different tactic is the full opposite, namely to invite all leaders of all sub-communities that may be present locally, and hope that this number may somehow represent the enigmatic community as a whole. In the Philippines, the rainforest was sometimes effectively defended by committees that did not only comprise the forestry ministry but also the leaders of the military, the church, the university, the farmers and others, however remote from the actual problem of forest protection.

Representation by Sampled Individuals

Obviously, citizens can enter participation processes without representing anything but themselves. We are concerned here however, with the question of whom to invite explicitly into a relatively small planning group. In that context, individual citizens should be selected as some form of sample from some category of significant stakeholders. Questions then are (1) what are the most significant categories? and (2) what size and type of sample will it be? Both of these questions are very difficult to answer, but a conscious handling of the dilemmas

[2] In the Nete basin in Belgium, 13 groups were carefully selected according to their relation to the river. But this did not at take into account that several participants wore multiple hats. For example, someone can be a Ministerial representative in the daytime and an environmental activist in the evening (Verhallen 2007). In the Netherlands, one member of the Consultative Group Inner City Deventer, a stakeholder forum discussing plans for bypasses of the river IJssel, was also a professional process facilitator in two other river interventions further upstream and downstream, as well as a local politician.

will already help much to avert risks of later under-representation and over-representation. Categories are notoriously fluid and hard to define, especially when we use the citizen sample to represent ‘difficult’ categories that are not represented already by the institutional entities (GO, NGO, interest organizations).

One way to go about this problem, as with the informal leaders, is to forget about all categories and seek to directly represent the diversity of the community as a whole. The sampling strategy then could be to ‘snowball’, *i.e.* to start out from a small number of community informants and through them identify named individuals that are thought to be typical for community sub-interests and sub-cultures.[3]

Representation on Paper

Participatory planning exercises, focusing as they do in informal and face to face interactions between participants, often neglect that stakeholders may also be represented on paper. This is especially feasible for stakes that fulfill two conditions, namely that they are (a) hard to get at the planning table and (b) relatively static and expressible in a formal manner. Many of those formal, on-paper expressions of stakes are in fact available to support multi-scale governance, in the form of regulations and standards issued by higher government levels (EU, national government etc.).

A well-known example in Dutch river management is the flood discharge condition put by the national river authority on river reaches. The rule says something like ”I do not care much about what you do to the floodplain here, as long as you create a lowering of the flood water level of so many centimeters compared to the present”. Recently, this rule has become accompanied by a portable hydraulic model that can calculate the flood level effect of floodplain measures in a matter of minutes. This has proven to be of great value for local floodplain design groups.

Other ‘paper representation of stakes’ may focus on general community aspects that may be surveyed and summarized, as were for instance public attitudes towards nature and river management that were represented on paper in a visioning exercise in Beunigen, the Netherlands (Swanenvleugel 2008). Finally, stakes that are difficult to represent may demand a specific effort to be translated into criteria and represented on paper. One example is the ‘playability’ of environments for children. Tremendous damage has been done all over Europe because bureaucracies think that what children need, if anything, are fixed seesaws and other playthings in stead of unregulated green space. In many public planning cases, playability should be discussed and laid down in a number of straightforward criteria. ‘Representations on paper’ will often require personal back-up in order to avoid that the paper becomes dead paper in the process. This may be a task of the participation facilitator or a specifically assigned person (e.g. the survey or playability researcher).

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[3] Such citizens may not realize fully in which ways they are attached to the problem at hand, e.g. the river. Unconventional ways have been found to engage people. The Centre for Sustainable Management of Resources at Radboud University, Netherlands, organized a poetry and song writing contest to enlist people’s attachment to the river Waal. As part of a visioning exercise, the same group used photographs to support people in discovering their floodplain landscape use and preferences (Swanenvleugel 2008). In Lodz, Poland, photography, film and amateur journalism facilitated by the European-funded SWITCH program for sustainable urban water management (www.switchurbanwater.eu) engaged people who would otherwise have remained passive, so that they could become participants.

Politicians and Business

Politicians and private market parties may sometimes be member of one of the above categories. A mayor or counselor may, for instance, be respected to such an extent that he may be one of the informal leaders. Likewise, a businessman may be detached from his own wallet enough to represent, say, the recreation interest as part of the common good. *As categories*, however, we include politics and business only last here. This has to do with models of democracy, as will be discussed below. We do not go into issues of representation here, assuming that the categories are usually organized enough to not create representation problems.

Whom to Actually Invite as Participant?

Up till now in the present section, we have discussed some ideas of how these significant stakes and stakeholders might be represented in a participatory setting. The next question to be addressed is: who will actually be invited to sit in? It is here that important issues of balance come to the fore. How may be group of participants be assembled such that not only each significant stake is represented individually, but also such that the group is a whole represents the problem situation as a whole? It is beyond our reach to offer a sophisticated ethics here. Obvious criteria, however, are:

- evenhandedness (equal thresholds of significance for being invited)
- active protection of the less articulate and voiceless
- avoidance of over-representation through different channels (e.g. the same stake represented by government and an interest organization and an informal leader)
- transparency (e.g. of invitation criteria and representation roles)
- consistency.

The last criterion needs some further explanation. We do so under the next two headings.

Consistency: Models of the Common Good

If we have a good sample of individual people with attribute X (e.g. ten people with the attribute of being a farmer), can then also the organization that groups people around that attribute (e.g. the farmers union) be stakeholder too? The answer depends much on the researcher's often implicit 'model of the common good'.

An individualistic (typically Anglo-Saxon) view on the individual versus the common good would tend to assume that the sum of individual goods equals the common good (hence either the farmers or the farmers union should be stakeholder), while a less individualistic vision, as often taken in France for instance, would tend to assume a more principled difference between the individual and the common good and would therefore tend to see both the farmers and their organization as stakeholders. Participation facilitators should be transparent on this point, and coherent, treating all common goods on the same footing.

Consistency: Models of Democracy

In the Habermasian ideal of democracy as a form of total deliberation, all parties communicate freely with each other, irrespective of their functions in society. In more traditional models of democracy, public life is divided into functional spheres with different roles and different internal rules. These spheres may be defined, for instance, as (a) politics, (b) the market and (c) civil society. Roles and rules then are, for instance, decision-making and one-man-one-vote, material production and competition, and immaterial production and voluntary deliberation, respectively. In this traditional model of democracy, participation is seen as a process in which the central coordination bodies of the state (roughly, the bureaucracy) reach out to civil society. Participation then excludes politics and business; it is an open space designed to give more voice to civil society.

Participatory processes can be coherently organized in both models of democracy. The high rule is that the game should be played coherently. In the traditional model, politics and business have their place outside the participation arena and should not be allowed halfway in. And the other way around in the Habermasian model, political actors and business actors are to be invited transparently and fully.[4]

STEP 4: SETTING A BALANCED SCENE

The preceding step on the road to from all stakeholders to balanced process has generated a list of desired participants in a planning project. Step 4 is centered at the period in which the project moves from its preparatory to its dynamic phase.

Two questions prevail at this point. First, overlooking the list of invitees, can we expect a sufficiently level playing field in the sense that all of them will have enough voice to be heard? Second, participants have been invited, but how do they respond to the invitation? Should other people be invited (or de-invited) as a response to who in fact shows up in the first planning sessions? Grouped under the heading of ‘setting a balanced scene’, these two issues will be discussed in this section.

Should the Intellectual Playing Field Be Leveled?

People may show up at participatory meetings, but are they knowledgeable? Mohapi (2003) tells the story of a man who had been to several stakeholder platform meetings on river management, and then when he finally dared to speak, asked the chairman to please explain what a stakeholder platform was. Is that problematic? *Should* people be knowledgeable? Let’s first clear two easy cases out of the way. Participants should be knowledgeable on the planning issue if and insofar (*a*) this is psychologically or socially conditional for them to participate at all, and (*b*) if they are supposed to represent a

[4] Out of the Habermasian model, forms of joint planning may ensue that de-emphasize civil society and with that, fall outside the rubric of public participation. In the successful model of WaalWeelde (Scholten 2009) , for instance, 14 municipalities work jointly on the design of room-for-river plans along the Waal river, represented by their political leaders (aldermen). Business parties participate actively in this process, but public participation is a relatively minor concern, even though creative efforts were made, e.g. through the internet.

knowledgeable group. But in other cases, how knowledgeable should participants be? If, say, a housewife with good verbalizing capacity sits in to represent housewives in a neighborhood planning process, how much should she know about neighborhood planning? Why worry about leveling the playing field in such a case? The more she comes to know about planning, the less representative she will be. A principled idea that the whole intellectual playing field should be leveled has a nicely emancipatory, democratic appeal. It tends to have an exclusivist result, however, because the time and funds to educate everybody are usually unavailable. The notorious effect then is to reinforce the common practice that experts first decide on the basic issues and then involve participants merely discuss details of implementation. In other words, the leveling of playing fields is a subtle matter that should not be dominated by our own intellectualist idea that everybody should be intellectualized, and neither by the experts’ ideas that experts know best. Too many river projects have been planned on ancient graves or holy sites, have sought to ‘improve’ a landscape that local people appreciate very much, and have ignored local knowledge and local ways of conflict resolution. This being said, a careful consideration of capacities and knowledge gaps can lead to the conclusion that the intellectual playing field should be leveled to at least some extent before balanced participation may be expected. In such cases, mutual learning can be set up as a joint process of experts and local people teaching each other (Warner and Simpungwe 2003; see also step 1 in the Joint Planning Approach at www.jointplanning.eu), in which experts and participants update each other on what the river means to them and what may be the major challenges and opportunities. Another method is to first create an ‘experts-free zone’ in which local people inform each other in the presence of the participation facilitator, which is intended to result in a more equivocal and self-confident local voice.

Participants behind the Curtain

Our definition of participants allows actors to participate in planning outside the public planning sphere. As Long (2001) puts it, participation is any action that social actors undertake to alter their conditions of living, whether or not it fits the box designed by initiators of public planning. Especially powerful affecting stakeholders may prefer not to been seen while participating in the planning process, and use other channels to influence planning outcomes or decision-making. In itself there is nothing sinister about the fact that, say, the landowner on whose land an intervention is planned, or the bank who will act as a guarantor, or a national conservation NGO use their contacts in the polity or bureaucracy to state their position. The legitimacy of such actions depends largely on the prevailing concept of democracy. In the Habermasian ideal of ‘total deliberation’, in which government and basically all stakeholders are supposed to ‘co-produce’ policies, the powers behind the scenes would need to be uncovered for balanced analysis and be drawn into the deliberation process. If this fails (as it usually does), participatory processes framed in the Habermasian ideal run a severe risk, since those who do participate may be led to believe that their preference will be translated into policy more or less automatically. Then finding that there are other, invisible circuits of decision-making is experienced as a breach of trust, which has led to many instances of conflict (Stirling 2008; Margerum 2002).[5] In the more traditional democracy model of separation of spheres, participation facilitators may say that what they are doing is trying to hear the voices of civil society and possibly help design a unified civil society plan or proposal, separate from the economic powers of the market and the decision-making responsibility of the political councils. Intermediate positions may be found as well (the most common of which is to involve as many affecting stakeholders as possible) but here again, consistency and transparency are of the essence.

Uninvited Guests

The major objective of the process of stakeholder analysis and participant invitation (Steps 1-3) is justifiable exclusion. A well-known phenomenon in practice is that of the uninvited actor showing up, claiming to be the representative of a stake unjustifiably ignored, or claiming to be the better representative of a non-ignored stake, or claiming nothing at all but being interested nevertheless. What to do? First of all, an impartial evaluation of the claim, of course. If no reasons for a rectification of invitations are found, one thing to do is to sit back and make the best of it, except when the additional presence turns out to be too disruptive.

Invited But No Show

As Moreyra and Wegerich (2006: 10) put it, “actors most of the times take what is at hand, transform and reshape interventions towards reaching their individual or collective goals”. Participation has a demand side, resulting in the invitations of Step 3. But it has a supply side too; invited participants must be willing to show up. Three reasons prevail why invited actors may decline to participate.

(1) Some stakeholders may prefer to be left out for principled reasons. The judiciary, for instance, guards its independence and shies away from any planning (lawmaking) involvement. Its position, roughly, is: “You bring us the conflict, then we’ll see”. Banks have the same rule of not wanting to sit in the planning seat: “You bring us the business plan, then we’ll see”. Usually, these positions are so well-known that these actors, even though they may be identified as strongly significant affecting stakeholders, are never invited to begin with.

(2) Other actors may prefer to stay out of the public participation process because they prefer to use other avenues of influence. Some of them will participate behind the scenes (see above). Others will in fact participate but prefer to do so on paper (see above; the position is: “go ahead, you know our rules”). Activist groups such as Greenpeace may feel that the give and take that comes with participatory negotiations is a risk for their public role. Finally, less powerful stakeholders may

[5] One such silent insider suddenly became visible when an alderman in Arnhem mentioned, during a public consultation on the planning of a neglected neighborhood, that the landowner would have veto power. The alderman's slip of the tongue made the participants lose faith in the whole exercise.

prefer to deploy the 'weapons of the weak' (Scott 1985), e.g. civil disobedience, boycott, delay, spreading rumors, and sometimes violence. In the evidence-based Rules Of Successful Practice designed by Margerum (2002), it is stated that giving all participants veto power over the plans to be made can significantly change the motivations of this category of actors, drawing them into the participation process and increasing the chances of implementation.

(3) Finally, actors may have no alternative avenue of influence and yet stay out of the participatory process. This could be called true non-participation and constitutes the most severe threat to the legitimacy of any public planning process. Factors that play a role lie at the benefit side as well as the cost side of participation. On the benefit side, an issue may simply not important or urgent enough for the actor, or the degree of influence that the actor perceives to possibly have may be too low. On the cost side, the actor may have difficulties to find the necessary time and may feel uncomfortable in the participatory situation, e.g. because of low verbalization capacity.[6]

Reflection and Remediation of Invited vs. Actual Participants

The preceding paragraphs have already supplied some thoughts and remedies concerning invited versus actual participants. The high rule is again that conscious reflections will already help much to take justifiable action. For instance, there is a great difference between an uninvited guest that only takes up an extra chair and some extra coffee, and an uninvited guest that dominates discussions. And as said about the problem of participants behind the scenes, much will hinge around a consistent and transparent handling of the model of democracy that the participatory process builds on. Participants that are invited but do not show up are the most intrinsically problematic group. Some general remedies do exist, however. One of them is to combine issues. People mail fail to show up first on neighborhood water issues, then again on neighborhood parks, then again on neighborhood crime and so on, but they may show up for integrated neighborhood renewal in which all issues are discussed. An allied solution is to avoid framing the problems from the start, and involve people from the beginning in the full agenda of deliberation, in a process of joint problem finding and joint problem framing. This may well lead to a broadening of issues, e.g. from narrow river management to broad river landscape planning. This in turn may open up new avenues for success also for the original river management challenge, because issues become tradable in package deals. River widening may for instance be combined with floodplain development that includes new nature and recreational areas (Swanenvleugel 2008).

STEP 5: MAINTAINING A BALANCED PROCESS

Step 4 has served to get a balanced process of participatory or joint planning in motion. The process will be a dynamic one, however, in which different phases, newly proposed

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[6] Other actors may feel the reverse in the latter respect; to have the opportunity to do one's citizen's duty or to express one's views is often felt as a joy in itself, irrespective of possible benefits (Warner and Verhallen 2005).

solutions and changing contexts require reconsiderations of what are significant stakes and who are desired participants. Although repeated reconsiderations cannot be called a ‘step’ in the literal sense, we have labeled it as Step 5 in our practical ethics. Windsor (1999) likens a participation process to a bus in which the number of seats does not match the number of passengers. He also maintains, however, that this problem can usually be solved in a balanced manner, using a mix of competition and cooperation, through which “some minimally satisfactory solution is found, such as standing in the aisle or changing seats periodically.” In the present section, we will meet both the “aisle” and “changing seats” phenomena.

Dynamic Invitation Connected to Project Phasing

Public planning processes go through phases such as problem identification, setting the rules of the game, visioning the outcomes, inventory of options for solutions, design of the draft plan, assessment of funding opportunities and constraints, and so on (see for instance the joint planning approach on www.jointplanning.eu). Considerations of whom to invite to participate may regard the process as a whole (see preceding section) but may also be left to vary according the phase. A wide representation of citizens, for instance, may be desirable in the visioning phase but a burden during funding searches and negotiations. The benefits of efficiency are obvious here, but so should be the risks to final effectiveness and legitimacy. In river planning in Tiel (the Netherlands), for instance, enthusiastic citizen groups felt cheated when excluded from later planning phases, without justifiable reasons known beforehand or even after the fact. One solution, in the vein of Windsor’s (1999) “standing in the aisle”, is to form a broad pool of participants and then include the ways of phase-wise selection out of this group in the discussion of the rules of the game very early in the planning cycle.

Dynamic Invitation Connected to Shifts in Problemshed

The features of the problemshed (problems, possible solutions and the connected stakeholders) may shift during the planning process. This may be due to gradual learning and reframing, e.g. on causes of the problem, its long-term impacts or the directions that solutions may go. It may also result from more drastic changes such as when issues are combined in order to enhance participation (see preceding section) or when solutions are proposed that have impacts on groups or regions there were hitherto not identified as stakeholders, e.g. when one community proposes to divert polluted streams of noisy flyroutes over the territories of another community. New solutions may also bring new affecting stakeholders into view, and the reverse may be true as well if solutions vanish from the scene.

Dynamic Invitation Connected to Actual Representation

People may have been invited as representing stake X, but will often turn out to in fact represent more or less than X. The NGO member standing in for future generations may turn out to be an ardent angler too, for instance.

The representative of farmers may turn out to be opposed to his own organic colleagues who in fact have no voice though this participant, and so on. Balanced representation may therefore require repair actions, adding and excluding participants along the way, in order to rectify over or under-representation.

Dynamics of Actual Participants

Irrespective of invitations, actual participants may drop in or out during the process. This may be due to 'forum shopping', reflecting the strategic behavior of actors, dipping their feet into a process here and there and assessing the odds of a achieving the desired outcome for them.

Stakeholders usually expect invitations even if they repeatedly decide not to come. They may find different planning phases more interesting than others, and appreciate the space to drop in again. Examples were observed in river visioning meetings in the Nete in Flanders and Dhünn/Wupper, Germany (Verhallen 2008 and Pochwyt 2008, respectively). In other instances, actors dropping in unexpectedly may be incited by social entrepreneurs who start to engage with a project at the interface between project and project-affected people (Long 2001).

A well-known phenomenon is that in late project phases, participants may be stirred into action – gatecrashing the party – when the project reaches their back garden. Reverse cases have been described as well, e.g. by Roth et al. (2008) when actors take the initiative at a very early project stage and spend great energies to stay in the lead.

The "acute sense of ethics" required of the participation facilitator (Decoster 2008) to keep the balance of representation in some justifiable equilibrium is especially essential during the heat of the participation process. The initial stakeholder analysis and round of invitations (Steps 1 to 4) can often be given some degree of systematic, open and multi-person thought. In the course of an ongoing participation process, however, these guarantees for balance are in much shorter supply. The facilitator will often be the only one making the balance decisions, and find himself under strong pressures to make the process a 'success' without too many questions asked.[7] Step 5 is the most difficult one for the participation facilitator in practice.

CONCLUSION

Most of the inspiring concepts and practices of relatively radical participatory planning and management (e.g. Borrini-Feyerabend et al. 2004) originate from experiences in developing countries or, to put is more precise, from contexts of morally and factually weak government, where creating space for power sharing and locally based co-management is both more necessary and more possible than in Western countries.

[7] As a result, any willing participant who speaks the language may find himself to be welcomed in an ongoing planning process, even if he doesn't represent anything. This has been the experience of one of the present authors on several occasions in participation processes on Dutch river works. After presenting himself, he was included without any discussion. This would have been impossible in the early (Step 1 to 4) phases of the participant selection.

In these Western societies, the division of roles between politics, bureaucracy, business and civil society requires much more careful consideration than the catch-all nets cast by participation facilitators in developing countries. Above this empirical South/West distinction hovers the theoretical tension between the Habermasian view of social actors pursuing consensus in open dialogue *vs* the Marxian view of conflict between vested interests and stakeholder groups that are framed into passivity by the ruling ideology. We have tried to find a flexible middle ground in these tensions – first by applying relatively objectifying (hence inclusive) ways to identify significant stakeholders and then by supplying a number of practicable concepts and rules to keep the balance between the stakeholder voices, including those of the voiceless, in the process. The justifiability of the representation decisions made during the analyses and discussions should not be sought first in the Habermasian, Marxian or any other theoretical vision but in the policy principles and the deeper, 'constitutional' rules of the society where the participatory planning process is acted out. Principles of equity (sustainability, protection of nature and the poor etc.) are often expressed with sufficient clarity at that level.

Overall, our method has been to develop the undifferentiated cloud of 'stakeholders' and 'participants' first into a few conceptual steps (significant stakeholders – framing reflection - invited stakeholders (participants) – actual participants – participants in the process) and then attach a number of categories, examples, reflections and sometimes a few rules to each of these. We hope that this system, though certainly not constituting any formal ethics, will serve to improve the quality and justifiability of the decisions of who is to represent what in participatory planning.

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Chapter 5

VOLGA RIVER: POLLUTION, WATER QUALITY, TOXIC CONTAMINATION AND FISH HEALTH

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ABSTRACT

The characteristic of current state of Volga river is given. Concentrations of organic and inorganic toxic substances in water are reported. Basic clinical and postmortem signs of fish intoxication are described; changes in the cellular structure of their organs and tissues, as well as disturbances in hemogenesis, developing under the effect of toxic agents, are characterized. The main disturbances to fish caused by the accumulation of microelements in their organs and tissues are also considered. Based on dose–effect dependencies calculated with respect to the total concentration of toxic substances, standardized to MPC, and fish health criteria, cases that exceed the critical levels of pollutants are demonstrated for the investigated river sections.

Keywords: Volga River, pollution, water quality, metal bioaccumulation, fish pathology.

INTRODUCTION

The *Volga* is the longest river of Europe. It flows through the western part of the Russia. It is Europe's longest river, with a length of 3,690 kilometres (2,293 miles), and forms the core of the largest river system in Europe. Because of the building of dams for hydroelectric power, the Volga is navigable for most of it's 2,293 km (3,692m) length. The Volga river basin which comprises 40% of the population of Russia, 45% of the country's industry and 50% of its agriculture. The biggest environmental problems stem from major industrial complexes, big dams, large cities and maintaining navigability. The problem being faced now is that this system and all of the associated infrastructure is inefficient. Domestic and industrial wastewaters, air-borne pollution of the catchment area, as well as non-sewerage effluents from settlement areas find their way to this water basin. Several studies have proved the contamination of water and accumulation of heavy metals, oil products, polycyclic aromatic hydrocarbons, polychlorirozak biphenyl's, dioxins, and other chemical compounds in bottom sediments (especially in the places of industrial effluents discharge) (Anthropogenic Impact…, 2003; State Report…, 2002, 2003). Water quality problems are most severe in European Russia, especially in the Volga Basin. Of all water withdrawn from natural sources in Russia, 33 percent comes from the Volga. About half of that water returns to the Volga as polluted discharge, accounting for 37 percent of the total volume of such material generated in Russia. The Volga's water does not meet the norms for drinking water and is unsuitable for fish farming or irrigation. The data on water contamination with toxic substances in the Volga River basin are discrepant due to several reasons (different time periods used for the analysis, non-coinciding sampling points, insufficient capacity of measurement instruments, etc.). In the late 1980s and early 1990s, numerous government committees were formed to clean up the Volga. Few of the resulting restorative programs have been implemented, however, and the Volga remains under ecological stress. Lately, due to the overall economic crisis and general industrial decay in the country, the input of pollutants to the Volga River has largely decreased. However, certain studies show that the level of water contamination remains high (Rozenberg, Krasnoshchekov, 1996; State Report…, 2002, 2003). One of the ecological consequences of water contamination with toxic substances in the Volga River basin and unsatisfactory water quality are very frequent cases of fish intoxication. Analysis of scientific papers dealing with the concentration of toxic substances and morbidity in fish in the Volga River basin shows that the situation is alarming. In 1965–1974, 334 cases of mass death of fish have been registered; in 1975–1985, 574 cases; 1986–1988 witnessed mass death of fish caused by 200 emergency and unit discharges of pollutants (Rozenberg, Krasnoshchekov, 1996).The attention of numerous researchers was attracted by very frequent incidences of myopathy (muscle exfoliation) and eggshell weakening in Volga–Caspian sturgeon. In spite of appreciable scientific and practical interest to the problem of the Volga River contamination with toxic elements and diseases in fish caused by such contamination, there are no system studies aimed at assessing the ecotoxicological situation within the investigated river basin.

Main objectives of study:

- To identify the modern levels of contamination of the Volga River water by toxic substances – metals and organic micropollutants;
- To study the accumulation of metals in fish as a consequence of increased concentration of metals in the water environment;
- To reveal the main pathological manifestations of chronic intoxication in fish of the Volga River;
- To assess the ecotoxicological consequences of increased toxic elements in water and the ecosystem health of Volga River on the basis of the pathological investigation of fish;
- To discuss critical levels of water pollution, and compare them with existing levels of pollution.

MATERIALS AND METHODS

In August and September of 2000–2002, comprehensive studies were carried out in 13 sections of the River Volga (Figure 1): the Ivankovskoe reservoir (I, II, III) of the Upper Volga; the Gorkovskoe (IV, V, VI) and Kuibyshevskoe reservoirs (VII, VIII) in the Middle Volga; and the course (IX, X, XII) and delta (XII, XIII) of the Lower Volga. Water was sampled for determination of the concentrations of toxic substances (metals and toxic organic compounds). Fish were examined to study their physiological state in order to reveal different forms of pathology and organ dysfunction. Water samples were always taken at the precise sites where fish were caught for examination. The bream *Abramis brama* (L.), the most widespread fish species in the Volga River basin, was used as a bioindicator. It is a benthic fish that does not make long-distance migrations, which enabled the collection of material for examination from limited sections of the river.

Figure 1. Location of the sections on the Volga River where the investigation was carried out.

Water chemistry. In total, 31 water samples were taken in 13 sections of the Volga River and reservoirs. Water samples were collected into Nalgen® Polyethylene bottles (1l and 60 ml). Bottles were cleaned in the laboratory and rinsed twice with lake water before sampling. After sampling, all samples were kept cool (approximately +4°C) in dark containers and were delivered to the laboratory within 1–3 days.

The analyses carried out on the water samples were as follows. The pH was measured using a Metrohm®pH-meter; conductivity (20°C) by Metrohm®-conductivity; alkalinity using the Gran titration method; and natural organic matter content by the Mn oxidation method. Microelement concentrations were determined using the atomic-absorption in graphite furnace method (GFAAS, "Perkin-Elmer - 5000" model, HGA-400, AAnalyst-800, Corp., Norwalk, USA). Hg was determined using atomic fluorescence (Fl, model Merlin®). Standard solutions with appropriate concentrations for each element were made from 1000 ppm AAS stock standards (Merk, Darmstadt, Germany). In addition, for determination of Hg, Mo, V, Se elemental analysis of the water was carried out by the inductively coupled plasma method using a "Plasma Quad 3" mass-spectrometer manufactured by Fisons Electronic Elemental Analysis (United Kingdom).

"Acidic" and "alkaline" extractions of the water samples (in glass bottles) were obtained with methylene chloride under field conditions. Concentrations of organic micropollutants in these extracts were determined by gas chromatography using a "QP-5000" chromate-mass-spectrometer manufactured by Shimadzu (Japan). The quality of the analytical results was repeatedly tested by intercomparisons during the course of the project (Hovind, 2000; 2002; Makinen, 2002).

An integrated impact dose is determined by summing the excess for each revile concentration of toxic compound to their $MPC_{fishery}$ as follows:

$I_{tox-1} = \sum(C_i / MPC_{fishery.})$

I_{tox} is the integrated toxicity index; C_i is concentration registered in water; $MPC_{fishery}$ is MPC for toxic substances accepted in Russia for fishery and aquatic life.

According to Russian rules of water protection, the water quality may be considered good if I_{tox} is no more than one ($0 < I_{tox-1} \leq 1$).Water quality may be considered good if I_{tox-1} is no more than one.

*Bioaccumulation***.** For determination of the metal content of the bodies of fish, subsamples from a minimum of five individual fish from every site were collected from the gills, liver, kidneys, muscle and skeleton. Samples of fish organs and tissues for metal analyses were dried to their constant weight at 105°C. Dry samples were prepared for analysis by wet digestion in ultrapure nitric acid (10 ml acid for 1 g of tissue). The content of Ni, Co, Cd, Cr, Mn, Pb, Cu, Zn, Al, Sr in fish was determined on an atomic absorption spectrometer, using a graphite furnace HGA-400. Duplicate analyses were used for the purpose of quality control.

In analyzing essential elements (Cu, Zn, Co) additional information about climatic variation along Volga River was also used, that is sums of annual temperature exceeding +10°C taken from climatic map.

Fish pathology. This was aimed at revealing the effects of toxic substances. Fish were studied at 13 river sections; the minimum number of fish observed was 50 of the same age (from 4+ to 6+ years old); all were free of internal parasites in the time period of the investigation (August and early September).

Blood samples are taken from live fish tail artery using methods described elsewhere. In the blood samples thus taken, hemoglobin concentration, erythrocyte sedimentation rate (ESR), erythrocyte and leukocyte concentration. Blood smear examination allows the analysis of red blood composition, differential blood count, and the detection of occurrence of pathologic blood corpuscles. (Ivanova, 1976; Krylov, 1980). Macrodiagnostics to determine fish health were carried out under field conditions. The clinical and pathological anatomical signs of intoxication and any abnormalities were documented on the basis of visual examination of the fish during the first hour after fishing.

In the process of visual examination, special attention paid to the following: the intensity of color, the state of pigment (cells–melanophores); the total amount of mucus on the fish body; the state of squama, opercula, oral cavity, anus; the cases of hyperemia, subcutaneous hemorrhages, sores, or hydremia of the body; deformation of skull and skeleton bones; the state of eye crystalline lens and cornea. When the opercula are opened, branchiae are examined, in particular, their color, the presence and the amount of mucus, the state of branchial petals (accretion, adhesion, dilatation, or thinning down). After the abdominal cavity is dissected, the state of fish muscles is studied (color, consistence, hemorrhages, attachment to bones), as well as the presence of exudate in the abdominal cavity, the amount of cavitary fat, its color and density. The topographic location of viscera (liver, kidneys, gonads, spleen, heart, stomach, intestines), their dimensions, color, density, edges, hemorrhages, zones of necrosis, etc. are studied. Mucous membranes of dissected stomach and intestines are examined, in addition to cerebrum, paying special attention to filling of vessels, their color and density. For more precise microdiagnostics, the organs of fish with overt signs of pathology were removed for histological analysis. Histological sections were prepared in the laboratory according to the standard method (Bucke, 1994). For satisfactory histological preparations only freshly killed fish were considered. Gills, kidneys, liver, and gonads were handled rapidly to prevent degenerative changes within the specimen. They were carefully dissected from the body, cut into blocks of $<1\ cm^3$ and placed in a fixative (Bouin's fluid). Histopathological alterations of organs were evaluated under a light microscope (450×). Diagnosis of disease was confirmed on the basis of histopathological observations. The percentage of sick fish in the stock of each local polluted zone was documented. Fish were detected at various stages of disease ranging from initially insignificant pathological organ changes to serious compromise of the organism. In the process of macrodiagnostics, three stages of disease can be identified (0 denotes healthy individuals):

(1) Low-level disturbance, not threatening the life of the fish;
(2) Medium-level disturbances, causing a critical state in the organism;
(3) Distinct signs of intoxication leading to inevitable death of the organism.

The overall index of morbidity in fish in a given zone of contamination can be presented as:

$$Z = (N_1 + 2N_2 + 3N_3) / N_{tot}.$$

Here Z is the morbidity index for fish, $0 \le Z \le 3$; N_1, N_2, and N_3 are the numbers of fish in the first, second, and third stages of the disease, respectively; and N_{tot} is the total number of fish examined in the local contamination zone, including healthy individuals. If none of the fish in a given body of water demonstrates any signs of intoxication, then Z = 0. The value of Z will increase with an increase in both the number of sick fish and the severity of their diseases.

Statistics. Statistical data processing was carried out using the regression analysis; the significance of correlation coefficients was determinated by *t*-criteria.

CHARACTERISTIC OF VOLGA BASIN AND ANTHROPOGENOUS LOADS

Geography and Hydrology. The Volga Basin comprises four geographical zones: the dense, marshy forests; the forest steppes; the steppes; and the semi-desert lowlands. It rises in the Valdai Hills of Russia, 225m above sea level north-west of Moscow. It also passes through a chain of small lakes. The Volga and its tributaries form the Volga river system, which drains an area of about 1.35 million square kilometres in Russia**.** The course of the Volga is divided into three parts: the upper; the middle; and the lower Volga. Starting as a small stream, it becomes a bigger river when it is joined by some of its tributaries. The major tributaries are the Oka, the Belaya, the Vyatka, and the Kama, each of which is longer than 1 000 km and has a catchment area exceeding 100 000 km^2.

The variation range of water discharge is great: from13420 m^3/c in flood time, 94 m^3/c in winter low water and 188 m^3/c in summer low water in the Upper Volga to 56500 m^3/c in flood time, 380 m^3/c in winter low water and 600 m^3/c in summer low water in the Lower Volga (Edelstein, 1998).

The Volga's flow is regulated by reservoirs. They accumulate about 70% Volga's flow. The morphometric characteristics of reservoirs are represented in the Table 1. The fierst two of them are included in the Upper Volga, the next tree – in the Middle Volga and the last two – in the Lower Volga. The Oka river falls after the Gorkovskoe reservoir and increases flow in two times, also the Kama river falls into the Kuibyshevskoe reservoir and increases flow in two times.

Table 1. The main characteristics of reservoirs in Volga river (compiled from (Edelstein, 1998))

Reservoir	Year of creation	Volume, km^3	Area, km^2	Maximum depth, km	Length, km	Volume water dropping through dam, km^3/year	Coefficient of water cycle, 1/year
Ivankovskoe	1937	1.1	327	19	120	9.2	7.9
Ribinskoe	1941	25.4	4550	28	112	30.1	1.4
Gorkovskoe	1955	8.8	1590	22	430	46.8	6.0
Cheboksarskoe	1981	4.6	1080	13	340	109.5	24.3
Kuibyshevskoe	1957	57.3	5900	41	510	234.9	4.2
Saratovskoe	1969	12.9	1831	33	336	230.6	19.1
Volgogradskoe	1961	31.4	3117	41	524	236.1	8.0

The Volga's major distributary, the Akhtuba, runs parallel to the main river on its way towards the Caspian Sea. Above Astrakhan, the Buzon River, another main distributary of the Volga, marks the start of the Volga Delta. The mouth of the river is situated on the Caspian Sea at 28m below sea level. As the Volga approaches the Caspian Sea it divides into a delta comprised of about 275 channels covering about 12000 km^2. The Volga Delta has a length of about 160 kilometres. It includes 555 channels and small streams. It is the largest estuary in Europe. It is the only place in Russia where pelicans, flamingoes, and lotuses may be found. The Volga freezes for most of its length for three months each year. Some of the biggest reservoirs in the world can be found along the river.

At the Caspian Sea the Volga is an important source of water for the sea and its famous sturgeon fishery. The Beluga sturgeon is the largest fish found in the Volga. But the water that flows into the Caspian has been used many times upstream by the factories and the farmers.

Anthropogenic impacts. Over half of Russia's industry is located within its drainage. The biggest environmental problems stem from major industrial complexes, big dams, large cities and maintaining navigability. Although the extensive development of the Volga has made a major contribution to the Soviet economy, it also has had adverse ecological consequences. The Volga basin is under pressure from human activities, industrial waste and chemical pollution being the most serious.

The industrial potential of the Volga basin is high and represent all industrial sectors. The most dangerous chemical and petrochemical industries come to the front, in Volga basin main capacities of oil processing (60%) and petrochemistry (70%) of Russia are concentrated. The central region specializes on production of plastic, chemical fibres, lacquers and paints, synthetic dyes, goods of home chemistry, the southern region – mineral fertilizers, caustic soda, polyvinylchloride and caprolactam. 14 enterprises produce pesticides. Should be noted that in 2000 9.3 million tons agricultural production were tested on pesticides and 47 thousand tons (or 0.35%) ones contented pesticides above permitted concentration (Rozenberg, 2009). The machine-building complex comes to the front too, but from 1991 to 2000 emissions of contaminations to the atmosphere have reduced in about 4.5 times and dump of contaminations to Volga – in about 3 times. The fuel and energy complex give maximum emissions to the atmosphere which have reduced in 1.7 times by 2000 (Rozenberg, 2009). The metallurgical complex don't be a ruling one in the Volga basin. The forest, woodworking and pulp and paper industries allocate in the north of Volga basin. Total and comparative dump of polluted water from all industrial source into Volga river are presented in the Table 2. Although the dump of polluted water has been reduced about on one third by 2000, load of wastes per unit of area and per one inhabitant has remained still high as compared with all Russian territory.

But most pollution in the Volga River watershed comes from nonpoint sources, or sources that are not easily traced back to a specific "point" like a wastewater treatment or industrial plant. In the Volga River watershed, nonpoint sources include areas used to land-apply manure, feedlots and pastures, and improperly connected or failing septic systems. Rainwater and snowmelt can wash waste from livestock (confined and pastured), pets, and wildlife into the river. To reduce the amount of fecal matter reaching the river, changes in waste and land management will be needed. It will take time to make these changes and to see the effects.

Table 2. Total and comparative dump of polluted water into Volga river (compiled from (Rozenberg, 2009))

Region	1991			2000		
	Total from all pollution source	Per unit of area, m^3/km^2	Per one inhabitant, m^3/person	Total from all pollution source	Per unit of area, m^3/km^2	Per one inhabitant, m^3/person
Upper Volga	901	3633	241	496	2000	133
Region	1991			2000		
	Total from all pollution source	Per unit of area, m^3/km^2	Per one inhabitant, m^3/person	Total from all pollution source	Per unit of area, m^3/km^2	Per one inhabitant, m^3/person
Middle Volga without Oka and Kama with Oka with Oka and Kama	4005 9844 11221	7783 13357 7484	218 238 219	2533 5997 7028	4923 8137 4688	138 145 137
Lower Volga	1234	4330	158	913	3202	116
All Volga Basin without Oka and Kama with Oka and Kama	6140 13356	5861 6572	205 213	3942 8437	3763 4151	132 134
Russia	27798	1628	16	20291	1188	12

The system of dams and reservoirs has blocked or severely curtailed access for such anadromous species as the beluga sturgeon (famous for the caviar made from its roe) and whitefish (*belorybitsa*), which live in the Caspian Sea but spawn in the Volga and other inflowing rivers, and it has fundamentally altered the habitat of the nearly 70 species of fish native to the river. These changes—along with pollution by industrial and municipal effluents.

As a result of climate change and an increase in the Volga's water temperature, fish like Kilka, a small, Caspian herring, have spread out. This shows that global warming does have a big impact on the river's ecosystem. More than 200 new species now live in the Volga permanently.

WATER CHEMISTRY AND CONCENTRATIONS OF TOXIC SUBSTANCES IN WATER

Water chemistry. Water chemistry not much differs along Volga river. pH values indicate a neutral reaction (the variation range is 6.2-8.0). Mineralization of water is low juding on electroconductivity which slightly increase on averige from 195 μSm/cm in the Upper Volga to 364 μSm/cm in the Lower Volga (Table 3).

Table 3. Concentration of microelements and their MPC values (μg/l) and also pH, conductivity (χ), calcium in water of Volga river (dash denotes the values below the detection limit; here and in tables 2, 3, 4, X is the average value; Min is the minimum value; Max is the maximum value)

Parameter,	Upper Volga		Middle Volga		Lower Volga		MP
element	X	Min-Max	X	Min-Max	X	Min-Max	C
pH	7.7	7.5-7.9	6.8	6.2-7.0	7.4	6.8-8.0	
χ, μSm/sm	195	165-268	255	226-280	364	357-387	
Ca, mg/l	29	25-40	36	32-38	35	32-37	
Hg	<0.05	-	<0.05	-	<0.05	-	0.01
Cd	0.13	0.08-0.20	0.13	<0.02-0.62	0.13	0.02-0.26	5
Pb	0.46	0.02-0.80	<0.02	-	1.72	1.00-3.20	6
Al	272.3	190-400	31.7	8.20-70.9	820.5	440-1480	300
Sr	101.1	85.0-120	190.8	95.7-289	521.3	469-568	400
Ni	1.78	1.20-2.50	1.72	<0.5-5.58	2.22	1.60-3.30	10
Mn	101.6	72.0-150	62.4	12.9-111	27.7	22.7-35.7	10
Zn	3.73	1.30-5.40	1.36	<1-5.11	5.58	2.60-8.70	10
Cu	2.14	1.20-3.80	2.21	0.94-5.68	1.70	1.30-2.30	1
Cr	0.83	0.29-1.70	0.75	0.49-1.18	0.60	0.47-0.75	70
Co	0.31	<0.2-0.60	0.16	0.14-0.19	0.60	0.30-1.30	10
As	2.8	1.8-4.7	1.0	0.7-1.4	1.8	1.2-3.2	50
Mo	0.20	0.16-0.23	0.62	0.43-0.82	0.43	0.38-0.49	1
V	1.36	0.98-1.65	1.21	0.85-1.71	2.35	1.95-2.59	1
Se	0.50	<0.5-0.55	0.84	<0.5-1.16	0.81	<0.5-1.17	2

The calcium and hydrocarbonates dominate in the ion composition, although in the southern region – Lover Volga concentration of more mobile aquatic migrants Na^+, Cl^- and also SO_4^{2-} increase till 15, 26 and 40 mg/l accordingly. The average values of permanganate oxidation and colour index are distributed as follows: in the Upper Volga – 18 mgO/l and 97°Pt-Co, in the Middle Volga – 10 mgO/l and 48°Pt-Co and in the Lower Volga – 15 mgO/l and <40°Pt-Co. That means contration of both easy oxidizable organic matter and total and bioavailable nutrients increase toward southern region.

Metals and metalloids. The microelement concentrations in the water were relatively low in the investigated river sections: the concentrations of Mo, Cd, Co, and Cr were less than 1 μg/l, those of Se and Pb varied from less than 1 to 1.7 μg/l, those of Ni, V, and Cu varied from less than 1 to 2.8 μg/l; the concentration of Zn varied from 1 to 6.2 μg/l; and that of As from 1 to 4.2 μg/l (Table 3). Relatively high concentrations of Mn and Sr were observed. The concentration of mercury did not exceed the accuracy of its determination using our technique (< 0.05 μg/l).

Relatively low concentrations of the investigated elements (especially Zn, Ni, Cd and Cu) can be explained by the absence of ferrous and non-ferrous metallurgical plants in the region under consideration, as well as by the overall decrease in the level of Volga River water contamination observed after the recent economic crises.

Comparison of the element concentrations in the Volga River with the respective "background" values for overland flow in European Russia (Petrukhin *et al.*, 1989; Burtseva *et al.*, 1991) showed that the concentration of As was higher than its "background" value in all the investigated areas; the concentrations of Ni and Cd exceeded the background level near the dam of the Kuibyshevskoe Reservoir; whereas the "background' concentrations of Cu and Se were exceeded in the central part of the Gorkovskoe Reservoir.

It is a well-known fact that zones of atmosphere and land contamination can be found within the catchment areas of the Kuibyshevskoe, Saratovskoe and Volgogradskoe reservoirs, as well as in the Lower Volga (Anthropogenic impacts, 2003). This probably explains the exceeding of "background" concentrations by such elements as V, Se, Pb, Ni, and Co. The concentration of Mn in the Ivankovskoe and Gorkovskoe reservoirs, as well as the concentrations of V and Cu in the Kuibyshevsoe Reservoir, the Lower Volga and the Volga River delta, were higher than the respective MPC values (List of Fishery standards, 1999), established for fishery water bodies.

Thus, the pattern of element concentration distribution within the investigated areas reflects, primarily, overall diffuse pollution, which is formed against the background of the natural geochemical input of microelements –and is mainly due to pollutant discharge by fuel and energy plants and general economic activity within the catchment area.

Toxic organic compounds. Dangerous organic substances include oil products, cyclohexane and cyclopentadiene and their derivatives, sebacic acid ether, xylene, phthalates, and dioxanes (Table 4). A high level of water contamination with alkyl derivatives of dioxane was revealed in the Gorkovskoe Reservoir.

Owing to large-scale application of polymer products, phthalates (used as plasticizing agents), and xylene (used for phthalic acid production) were observed in all the investigated sections of the Volga River (especially in the Gorkovskoe Reservoir and the delta downstream of Astrakhan). Dibutyl phthalate, whose concentration in the water varied from 1.3 to 55.7 μg/l (its MPC for fisheries is only 1 μg/l), deserves special attention.

The MPC for oil products was not exceeded in the investigated sections of the Volga River except near the dam of the Gorkovskoe Reservoir, which is probably affected by effluents from petrochemical enterprises located upstream of the Gorkovskoe Reservoir near Yaroslavl city.

Dangerous substances such as chlororganic compounds were not found in the investigated sites, which could be explained by their absence from the water or by their low concentrations (not exceeding the "sensitivity threshold" of the applied method). Analysis of scientific papers and data collected by the Hydrometeorological Service of Russia has shown that, in individual samples of Volga water, the concentrations of certain dangerous substances (such as DDT, DDE, alpha-hexachloran, and gamma-hexachloran) exceed the MPC. In 2002, these substances were also found in the Kuibyshevskoe Reservoir (State Report, 2003).

Thus, both organic and inorganic pollutants, for which the Toxicological Harmfulness Value has been established, are found in the Volga River water.

Table 4. Concentration of toxic organic substances and their MPC values (μg/l) in water of Volga river

Toxic organic substances	Upper Volga		Middle Volga		Lower Volga		MPC
	X	Min-Max	X	Min-Max	X	Min-Max	
Hydrocarbons of oil products:							
Alkanes	12.0	1.45-19.3	31.4	6.2-114	10.0	0.9-17.7	
Alkenes	0.1	0-0.4	1.6	0.7-4.0	0.5	0.2-1.8	
Total	12.1	1.45-19.7	33.0	6.2-118	10.5	0.9-19.5	50
Monatomic saturated alcohols	0.8	0-2.0	1.3	0-5.1	1.1	0.1-4.6	500
Ethers of carboxylic acids:							
Dioctyl cebacate	3.9	0-11.6	0	0	0	0	1
Toxic organic substances	Upper Volga		Middle Volga		Lower Volga		MPC
Carbocyclic compounds:							
Cyclohexane and its derivatives	0	0	0	0	0.1	0-0.8	10
Cyclopentadiene and its derivatives	0	0	0	0	0.4	0-1.0	10
Aromatic compounds:							
Xylene	0.3	0-1.0	1.3	0-2.5	0.5	0.2-1.3	50
Isopropyl benzene	0	0	0	0	0.1	0-02	100
Orthophthalic acids ethers:							
Dibutyl phthalate	2.8	1.3-4.5	25.2	5.7-55.7	32.1	9.6-44.5	1
Dioctyl phthalate	11	4.0-17.7	18.6	0-47.2	1.1	0.5-2.3	10
Heterocyclic compounds:							
Derivatives of 1,3-dioxane	0	0	27.6	2.0-81.7	0	0	10
Sum of chlororganic pesticides (DDT, DDE, α-hexachloran, γ-hexachloran)	0	0	0.01	0-0.04	0	0	0.01

METALS IN FISH AS A REFLECTION OF WATER POLLUTION

The concentrations of metals in fish can reflect levels of pollution more accurately than the indices of contaminant content in water (Moor, Rammamoorthty, 1983; Spry, Wiener, 1991; Moiseenko, Kudryavtseva, 2002). A group of non-essential elements (Hg, Cd, and Pb) is most dangerous for living organisms. The concentrations of these metals in the environment is increasing steadily (Dirilgen, Doğan; 2002; Friedmann *et al.*, 2002; Gochfeld, 2003; Moiseenko *et al.*, 2006*)*.

Mercury. The concentration of Hg in bream organs and tissues varied from less than 0.001 to 0.127 μg/g dry weight (Table 5). This metal accumulates most intensely in the liver, kidneys, and muscles, as confirmed by data presented in the scientific literature (Moore, Rammamoorthty, 1983; Friedmann *et al.*, 2002; Gochfeld, 2003). The highest concentrations of Hg were revealed in bream caught in several sections of the Middle Volga, which is subject to the heaviest anthropogenic load. Comparison of the data obtained by the authors with those from other scientific papers showed that the limits within which the concentration of Hg in bream muscles and liver can vary are comparable with those determined for Lake Balaton (Farkas *et al.*, 2003) and certain water bodies in the Czech Republic (Svobodova *et* *al.*, 1999). Similar values were cited for certain freshwater and sea fish inhabiting water bodies of the USA (Watras *et al.*, 1998).

Table 5. Concentration of microelements in the organs and tissues of the investigated breams (μg per 1 g of dry weight). Here S_x is the standard error

	Upper Volga		Middle Volga		Lower Volga	
Element	$X \pm S_x$	Min-Max	$X \pm S_x$	Min-Max	$X \pm S_x$	Min-Max
1	2	3	4	5	6	7
gills						
Hg	0,011±0,001	0,005-0,022	0,011±0,002	0,004-0,035	0,005±0,001	0,001-0,015
Cd	0,05±0,01	0,01-0,19	0,01±0,00	0,01-0,04	0,20±0,03	0,05-0,42
Pb	0,33±0,07	0,05-0,46	0,04±0,01	<0,01-0,16	0,07±0,01	<0,01-0,17
Al	52,9±6,7	14,7-110,3	9,38±0,91	4,38-17,3	106,6±15,2	34,6-199
Sr	128±4	96-163	299±24	167-458	790±46	394-1095
Ni	0,23±0,03	0,03-0,48	1,32±0,32	0,30-4,79	1,18±0,14	0,40-2,36
Mn	89,0±5,1	52-134	61,5±3,4	45,0-87,2	28,9±1,4	18,0-39,0
Zn	82,0±1,7	67,4-94,2	81,2±1,8	72,3-90,6	81,4±1,5	75,5-94,4
Cu	2,31±0,07	1,60-2,82	1,95±0,08	1,31-2,63	4,22±0,29	2,91-6,99
Cr	0,28±0,04	0,04-0,66	0,14±0,01	0,06-0,25	0,70±0,14	0,21-2,08
Co	0,10±0,02	<0,01-0,30	0,55±0,05	0,17-0,83	0,32±0,07	<0,01-0,98
muscles						
Hg	0,019±0,003	<0,001-0,041	0,049±0,005	0,023-0,092	0,031±0,004	0,014-0,066
Cd	0,03±0,01	<0,01-0,09	<0,01	<0,01-0,02	0,04±0,02	<0,01-0,22
Pb	0,07±0,02	<0,01-0,18	0,06±0,01	<0,01-0,13	0,02±0,00	<0,01-0,06
Al	4,17±0,54	0,91-7,16	1,54±0,18	0,65-2,79	2,38±0,27	1,06-4,11
Mn	4,10±0,38	1,02-6,89	3,12±0,19	1,98-4,44	1,24±0,15	0,33-2,43
Ni	0,13±0,03	<0,01-0,51	0,22±0,02	0,10-0,43	0,26±0,03	0,12-0,60
Sr	4,63±0,27	2,79-6,21	9,74±1,08	5,15-19,3	16,2±3,28	1,40-42,2
Zn	17,5±0,50	14,1-21,7	20,0±0,9	15,3-25,9	25,0±1,2	17,1-32,5
Cu	0,67±0,03	0,40-0,91	0,75±0,07	0,34-1,24	1,07±0,07	0,67-1,61
Cr	0,10±0,02	<0,01-0,34	0,04±0,01	<0,01-0,10	0,18±0,03	0,05-0,48
Co	0,03±0,01	<0,01-0,10	0,18±0,02	0,01-0,28	0,15±0,02	0,02-0,30
liver						
Hg	0,053±0,005	0,027-0,086	0,048±0,009	0,011-0,127	0,054±0,010	0,001-0,103
Cd	0,25±0,04	<0,01-0,67	0,26±0,04	0,11-0,69	0,35±0,06	0,01-0,59
	Upper Volga		Middle Volga		Lower Volga	
Element	$X \pm S_x$	Min-Max	$X \pm S_x$	Min-Max	$X \pm S_x$	Min-Max
1	2	3	4	5	6	7
liver						
Pb	0,25±0,07	0,01-0,75	0,19±0,04	0,04-0,66	0,06±0,01	0,01-0,15
Al	6,62±1,01	1,59-15,7	6,55±1,02	1,86-14,1	6,06±0,92	3,05-14,3
Sr	0,53±0,08	0,16-1,53	0,82±0,12	0,05-1,66	1,81±0,55	0,46-8,03
Ni	0,20±0,04	<0,01-0,56	0,19±0,03	0,03-0,41	0,33±0,04	0,12-0,63
Mn	8,27±0,34	5,72-11,1	6,93±0,38	4,97-8,94	6,23±0,44	3,56-9,00
Zn	99,7±5,4	51-143	79,6±6,4	51-140	110±10	58-205
Cu	46,7±4,4	7,4-79,8	35,5±7,5	10,3-114,2	89,3±10,4	29,8-155
Cr	0,19±0,05	<0,01-0,73	0,07±0,01	0,02-0,16	0,20±0,04	0,02-0,50
Co	0,09±0,01	<0,01-0,27	0,21±0,03	0,06-0,43	0,16±0,02	<0,01-0,32

Cadmium. The literature cites a high degree of Cd accumulation in living organisms, which is indicative of environmental pollution on local and regional scales (Conto-Cinier *et al.*, 1997). The most intense accumulation of Cd in all the physiological systems of fish was observed for those inhabiting the Lower Volga (Table 5). The accumulation of Cd in fish muscles testifies to long-term pollution of the body of water with this metal (McGeer *et al.*, 2000). The maximum Cd concentration (up to 5.66 μg/g dry weight) was recorded in the kidney. Unfortunately, there are few studies devoted to the analysis of accumulation of this metal in fish kidneys. The concentration of Cd in fish kidneys closely correlates with its concentration in other systems of the fish organism, such as the liver (r = 0.78, $p<0.005$), skeleton (r = 0.71, $p<0.01$), and gills (r = 0.56, $p<0.1$), which demonstrates penetration of Cd into the fish from contaminated water.

Lead. The maximum concentration of Pb in bream was observed in the Upper and Middle Volga (Table 5). Pb is most intensely accumulated by the kidneys, liver, and muscle (the respective concentrations were up to 1.3, up to 0.75, and up to 0.18 μg/g dry weight). It is difficult to explain why, under conditions of a higher concentration of Pb in the Lower Volga water, the maximum accumulation of this element was observed in fish inhabiting the Upper and Middle Volga. This is probably a manifestation of the cumulative effect of river contamination in previous years. In Lake Balaton, the concentration of Pb in bream muscles was higher: 1.6 μg/g dry weight (Farkas *et al.*, 2003). *Aluminum.* According to data from Rosseland *et al.* (1990), a high concentration of Al in the environment (both in dissolved and suspended forms) ensures its intense accumulation in fish (especially in the gills). The maximum concentration of Al in the water and in organs and tissues of bream was observed in the Lower Volga (Tables 3. and 5). Accumulation of Al could be traced in all the body systems of fish, but the highest concentrations of this metal were observed in the gills and skeleton. Bioaccumulation of Al in these organs is demonstrated by the following regression equations:

$Al_{gills} = 0.072\ Al_{water} + 22.4$, r = 0.90, $p<0.001$;
$Al_{skeleton} = 0.007\ Al_{water} + 7.19$, r = 0.74, $p<0.01$.

The close dependence of Al in fish gills on Al in water can be explained by the fact that, in the process of breathing, water is filtered through the gills of the fish, and Al settles onto the gill surface. Coagulation of Al on the surface of the gill epithelium, in addition to its inclusion in epithelial cells, has been demonstrated (Rosseland *et al.*, 1990). *Strontium.* This element participates in metabolic processes with Ca. Being more labile and active, Sr gradually disturbs the normal calcification of the skeleton and causes pathological disturbances in bone tissue (Chowdhuury *et al.*, 2000). The highest concentration of Sr in bream inhabiting the Volga basin was found in the Lower Volga sections, where the concentration of Sr in the water was maximal (Table 5), reaching 1500 μg/g dry weight in fish skeleton and 1100 μg/g dry weight in the gills. Sr accumulates not only in fish bones but also in fish muscles, liver, and kidneys. The dependence of the Sr content of bream organs and tissues on its concentration in water can be approximated by the following equations:

$Sr_{gills} = 1.52\ Sr_{water} - 25.0$, r = 0.99, $p<0.001$;
$Sr_{muscles} = 0.026\ Sr_{water} + 2.98$, r = 0.74, $p<0.01$;
$Sr_{liver} = 0.003\ Sr_{water} + 0.191$, r = 0.79, $p<0.005$;
$Sr_{kidneys} = 0.006\ Sr_{water} + 0.672$, r = 0.95, $p<0.001$;
$Sr_{skeleton} = 1.94\ Sr_{water} - 73.6$, r = 0.98, $p<0.001$.

It should be emphasized that the Sr/Ca ratio in water varied along the river course; it was 1/289 in the Upper Volga, 1/186 in the Middle Volga, and 1/66 in the Lower Volga water. In addition to the increase in the absolute value of Sr concentration in the water from the Upper to the Lower Volga, its relative concentration in fish organisms increased even more, which testifies to the replacement of Ca by Sr in bream bones. For example, the Sr/Ca ratio in bream gills was 1/516 in the Upper; 1/266 in the Middle; and 1/83 in the Lower Volga. The respective values for bream skeleton were 1/798, 1/384, and 1/116. Thus, Sr features a high bioaccumulation capacity. *Nickel.* The concentration of this metal in the muscles and liver of bream inhabiting the Volga River did not exceed 0.60 μg/g dry weight; for kidneys, the value was somewhat higher (Table 5). Ni accumulates intensely in fish, mainly in the gills and kidneys (Moiseenko, Kudryatseva, 2002). The ability of this metal to accumulate is confirmed by the regression dependencies between the concentration of Ni in the water and in Volga bream organs and tissues (with the exception of the skeleton):

$Ni_{gills} = 0.343\ Ni_{water} + 0.197$, r = 0.68, $p<0.025$;
$Ni_{muscles} = 0.086\ Ni_{water} + 0.058$, r = 0.82, $p<0.005$;
$Ni_{liver} = 0.051\ Ni_{water} + 0.155$, r = 0.73, $p<0.01$;
$Ni_{kidneys} = 0.206\ Ni_{water} + 0.477$, r = 0.76, $p<0.01$.

Thus, the accumulation of Ni in fish organisms depends on its concentration in the water, but the concentration of this metal in the water and fish of the Volga River is low. *Manganese.* Mn is usually considered to be of low toxicity. According to Musibono and Day (1999), Mn reduces the toxicity of such elements as Cu and Al, i.e. Mn possesses antagonistic properties in multicomponent water contamination. Mn is irregularly distributed in the Volga River water: the concentration of Mn in the Upper Volga was much higher than in the Lower. The concentration of Mn in fish organisms changes similarly: the most intense accumulation of Mn was observed in bream inhabiting the Upper Volga, and the maximum amount of this metal was found in the bream gills and skeleton (Table 5). A significant correlation was revealed between the concentration of Mn in fish organisms and in the respective water (for gills, $r = 0.68$, $p<0.025$; for muscles, $r = 0.67$, $p<0.025$; for liver, $r = 0.61$, $p<0.05$). A close correlation was also revealed between the values of Mn concentration in different tissues and organs of the same fish individual, which testifies to synchronous bioaccumulation of this metal depending on its concentration in the water. *Zinc, chromium, copper, and cobalt.* These are essential elements. No distinct patterns could be traced in the distributions of Zn, Cr, and Co concentrations in the Volga River water. As a rule, Cu, Zn and Co accumulate in fish liver, where active metabolic processes take place. Their maximum concentration was found in the livers of Lower Volga bream (Table 5). It is well established that the rate of metabolic processes in fish is determined by the ambient temperature. If the concentrations of microelements in the water are similar, the rate of their bioaccumulation can depend on the temperature conditions. The availability of essential elements to functionally vital organs of bream inhabiting the Lower Volga is probably affected by the intensification of metabolic processes in warmer water. A correlation was found between the concentrations of Cu, Zn, and Co in fish muscles and the sum of annual temperatures exceeding 10°C (Figure 2), whereas no such correlation could be established between the concentrations of the above elements in the fish organs and in water.

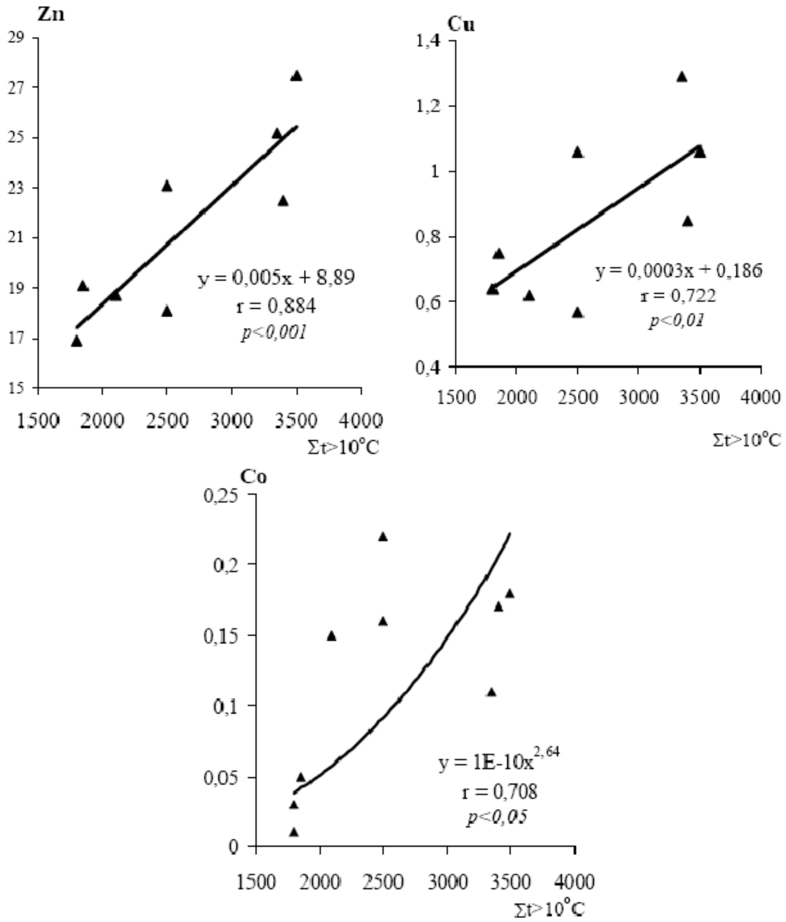

Figure 2. Dependence of essential element concentrations in muscles (μg per 1 g of dry weight) on the sum of temperature values exceeding +10oC (Σt>10oC).

Multimetal penetration. The accumulation of microelements in fish organs and tissues causes microelementoses, i.e. changes in the ratio of microelement concentrations in fish organs and tissues. A high correlation was established between the concentrations of certain elements in the bream organs, which testifies to the effect of pollution on the increase in the concentrations of the investigated microelements in fish gills:

Sr→(r = 0.87, $p<0.001$)←Cd→(r = 0.96, $p<0.001$)←Al→(r = 0.78, $p<0.005$)←Cr,

and in fish kidneys:

Cr→(r = 0.53, $p<0.1$)←Sr→(r = 0.75, $p<0.01$)←Cd→(r = 0.55, $p<0.1$)←Ni→(r = 0.72, , $p<0.05$)←Co.

This group of elements accumulates mainly in fish inhabiting the Lower Volga. Accumulation in fish liver of Hg→(r = 0.61)←Zn is observed in the Middle Volga, whereas accumulation of Mn→(r = 0.59)←Pb in fish gills and liver is typical of the Upper Volga.

Bone tissues in the investigated fish demonstrated a high degree of correlation between the concentrations of the following elements:

Pb→(r = 0.66, $p<0.025$)←Hg → (r = 0.79, $p<0.005$)←Ni.
↑
Cd→(r = 0.79, $p<0.005$)←Cu→(r = 0.87, $p<0.0015$)←Sr.
↓
(r = 0.76, $p<0.01$)←Zn.

The correlations established between the concentrations of essential and non-essential microelements prove their joint penetration into the fish organism as a result of multimetal pollution. Based on the analysis of the element distributions in bream and the joint penetration of certain microelements into them, Sr-Cd-Al-Cr-Ni anthropogenic hydrogeoformation can be singled out in the Lower Volga; Hg-Zn hydrogeoformation in the Middle Volga; and Mn-Pb hydrogeoformation in the Upper Volga.

FISH PATHOLOGY

Various deviations from the physiological norm were found in all the fish of investigated river sections.

Gills. In some cases, the gills were pale (their normal color is scarlet) with a clearly distinct anemic ring along the gill arc. The largest number of fish with an anemic ring was caught in the Gorkovskoe Reservoir and in the Lower Volga (downstream of Astrakhan).

Epithelium desquamation in secondary lamellae (Figure 3.c), swelling of the distal parts of filaments, and shortening, curvature, and fusion of secondary lamellae (Figure 3.b) were observed, which resulted in the transformation of the rigidly structured gill into an unstructured mass, with the distal filament alone still functioning. Congestive phenomena (stasis) were found in most of the respiratory lamellae, which is related to the violation of capillary conductivity. Vast hemorrhages were observed between filaments and secondary lamellae (Figure 3.d). In certain filaments, the secondary lamellae were completely destroyed. Extensive lammelar hypertrophy with some proliferation from the bases of the secondary lamellae was recorded.

Liver. Changes in the liver color, dimensions, and texture were observed. All the bream caught in different river sections had increased loose-textured liver with color varying from a mosaic light-brown to pale yellow. In some cases, the liver was liquified; it had clearly distinct parts of necrosis or pronounced signs of atrophy. All the examined fish demonstrated signs of liver disease of differing degrees of severity. Frequent visible disturbances of this organ were typical of fish caught in the Gorkovskoe (up to 92.6% of the fish) and Kuibyshevskoe (up to 54.5% of the fish) reservoirs, as well as in certain sections of the Lower Volga (Table 4). Morphological and functional changes in the liver manifested themselves in the form of lipoid dystrophy (Figure 4.b) and hydropic dystrophy (Figure 4.c), which are symptoms of progressive hepatopathy. In the case of intensified intoxication, lipid and hydropic dystrophy of hepatocytes were often found. Hydropic dystrophy is a variation of protein dystrophy and is related to the disturbance of protein and water exchange. In this case, the permeability of cell membranes increases, vacuoles appear in the cytoplasm due to water ingress, the cellular organelles are destroyed, while the cell itself becomes filled with water and dies. In the case of lipoid dystrophy, fat occlusions, which almost completely fill the cells, appear in the hepatocytes. Diffuse disruptions of bream liver, accompanied by disturbances in the morphological structure of liver lobules and pronounced necrosis of liver tissue, were also diagnosed. Mechanisms of lipoid and protein dystrophy development are similar. Frequently, they develop under the conditions of the organism intoxication or accompany hypoxia. In some microscopic sections, complete necrosis (not that of a "hotbed" character) of the liver tissue was observed (Figure 4.d). Interstitial proliferative inflammation related to hepatocyte necrosis and the appearance of inflammation infiltrates were also diagnosed. In the process of their development, the cells of the infiltrates transform into collagenous fibres of connective tissue. As a result, a thick connective-tissue capsule can appear around the zone of necrosis. Such progressive necrosis and structural reorganization of the tissue can contribute to post-necrotic hepatic cirrhosis, leading, in turn, to hepatic failure. Vast zones of parenchymal hemorrhage, destruction of blood corpuscles and blood vessel walls, as well as proliferation of connective tissue around the blood vessels, were revealed.

Signs of chronic congestive hyperemia in liver veins were found. They testify to varicose veins and capillaries, a decrease in blood pressure, and blood flow deceleration. As a result, the supply of the tissues with blood becomes disturbed, and tissue hypoxia occurs. All these processes taking place together can lead to congestive edema. Disturbances revealed in the liver cell structure entail the development of first sclerosis, and then cirrhosis.

Kidneys. The largest number of fish with pathological disturbances in the kidneys was caught in certain areas of the Gorkovskoe and Kuibyshevskoe reservoirs (Table 4). Pathological disturbances in the kidney tissue manifested themselves in fibrosis, where vast connective-tissue accretions substituted zones of necrosis in the canaliculi and interstitial tissue. In medicine, similar histopathology is typical of interstitial nephritis (fibroelastosis). In the connective tissue between the kidney canaliculi, pronounced interstitial inflammation (a diffuse infiltrate composed of blood cells) was observed (Figure 5.f). Signs of congestive hyperemia in the veins were revealed. Severe degeneration of adipose tissue was also diagnosed (Figure 5.d). In this case, the adipose tissue had a clearly formed structure; lipocytes were organized in groups ("lobules", separated from each other by membranes with blood vessels). The following disturbances were found: destruction of lymphoid tissue (Figure 5.b); proliferative inflammation (Figure 5.c), with zones of necrosis surrounded by thick connective-tissue capsules, separating the disturbed zone from normally functioning tissue and preventing the proliferation of pathology; and the occurrence of interstitial substances in the kidney parenchyma (Figure 5.b), causing compression of healthy tissue, which, in the long run, can lead to the organ atrophy.

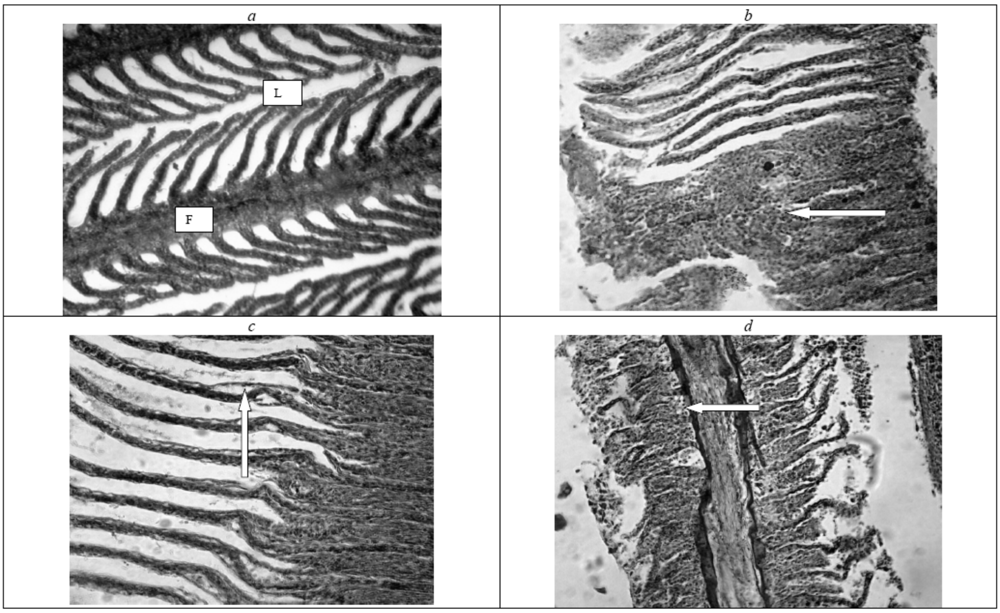

Figure 3. Pathological changes in the gills of bream (arrowed): a – normal structure (F – filament, L - lamellae), x160; b – extensive lamellar hyperplasia with fusion of secondary lamellae, x320; c – separation of epidermis at base of secondary lamellae, x320; d – hemorrhage, x160.

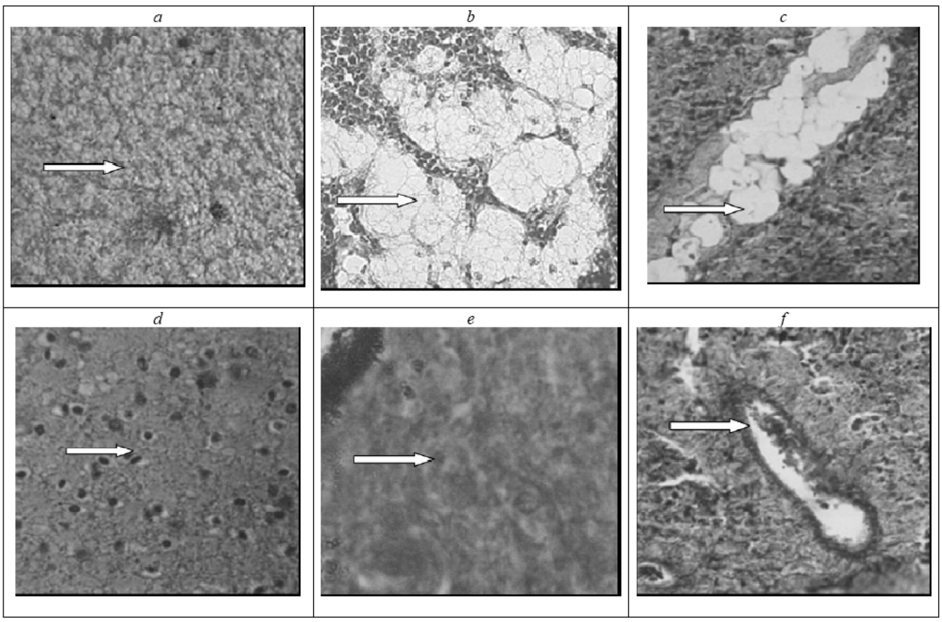

Figure 4. Pathological changes in the liver of bream (arrowed), x320: a – normal structure; b – lipoid dystrophy; c – hydropic dystrophy; d – karyopycnosis and necrosis of hepatocytes; e – inflammation; f – breakdown of blood cells.

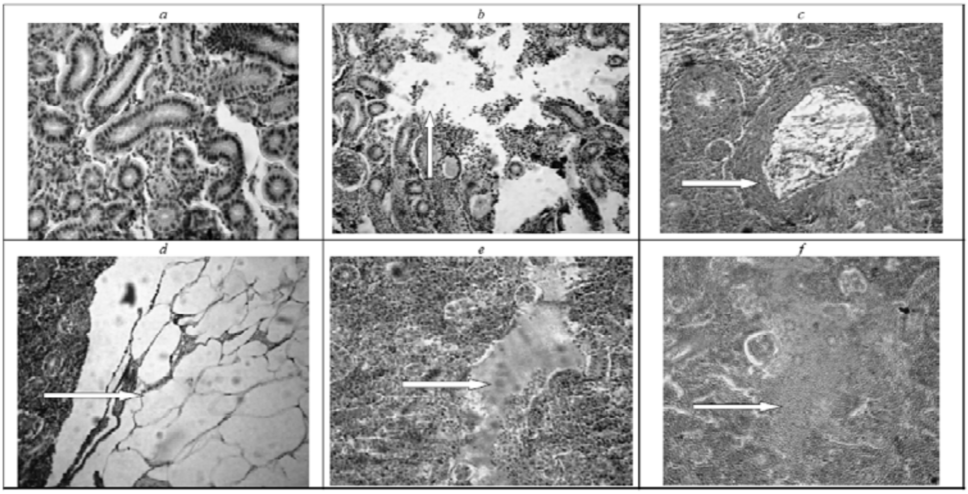

Figure 5. Pathological changes in kidneys of bream (arrowed), x320: a – normal structure; b – necrosis of the hematopoietic tissue; c – proliferative inflammation with fibrosis of the hematopoietic tissue; d – lipoid degeneration; e – interstitial inflammation; f – hemorrhage.

Gonads. At the time of the examination, gonads were at stage II–III of development. In some cases, their form and texture were abnormal. Their growth was uneven. Some parts of gonads were replaced by nonfunctional connective tissue. Twisting of the gonads was typical, mostly in males. In females, uneven development of fish eggs was observed.

Hematology. Pathology developed simultaneously with disturbances in vitally important organs of the fish. The "norm" of hematological indices is different for each fish species. According to the data presented by Zhiteneva *et al.* (1989), the concentration of hemoglobin in the blood of healthy bream varies from 92.0 to 101.0 g/l. A 15% to 30% decrease in hemoglobin concentration is a signal of fish disease, which can be caused by both invasive and toxic agents. For bream inhabiting the Volga River basin, a value of 90 g/l is adopted as the lower boundary of the "norm" of natural variability in hemoglobin concentration. The largest number of fish whose hemoglobin concentration was lower than the norm, was caught in a certain site of the Lower Volga and in the Gorkii Reservoir.

Toxic substances affect not only hemoglobin concentration but also change the leukogram and red blood cell composition (Ivanova, 1976; Zhiteneva *et al.*, 1989). Studies have shown that, in different sites of the Volga River, the ratio between different forms of blood cells of bream changes. The highest percentage of immature forms of erythrocytes was found in blood smears of fish caught in the Lower Volga, which is in agreement with the low hemoglobin concentration in the blood. Changes in the leukogram of the bream manifested themselves in an increase in the relative amount of neutrophils and monocytes, especially in fish from certain sections of the Lower Volga and the Gorkovskoe Reservoir (Table 6). In the blood smears, different pathological forms of erythrocytes (lacy erythrocytes, poikilocythemia, vacuolization of the cytoplasm, pycnosis of the cell nuclei, amitosis of the cell nuclei, etc.) were found. The changes revealed in hematological parameters of the examined fish confirm the development of toxicoses in fish inhabiting the Volga River basin.

Thus, fishes caught in the Volga basin had visible clinic and postmortem symptoms of intoxication. The degree of disturbances in their organs varied from hardly visible to pronounced deep degenerative changes, increasing the risk of death of the individual.

Table 6. Characteristics of the physiological state of fishes caught in the Volga river

Parameter	Upper Volga		Middle Volga		Lower Volga	
	X	Min-Max	X	Min-Max	X	Min-Max
Z	1.54	1.33-1.71	1.97	1.71-2.11	1.45	1.00-1.74
Percentage of the fishes demonstrating second and third stages of the disease	44.3	37.7-52.4	72.9	53.6-85.2	41.8	20.0-64.1
Percentage of the fishes demonstrating pathological disturbances in the liver	56.5	41.0-64.3	68.4	46.4-92.6	29.5	18.2-44.4
Percentage of the fishes demonstrating pathological disturbances in the kidneys	25.4	21.4-28.6	59.9	32.1-80.0	9.4	0-25.6
Hemoglobin (Hb), mg/l	103	81-124	96	52-126	88	56-122
Percentage of the fishes with Hb not exceeding 90 mg/l	15.2	0-37.5	22.0	11.8-40.0	54.3	9.1-80.0
Leucocytes:						
lymphocytes, %	87.5	82-93	85.8	68-94	64.7	29-93
monocytes, %	1.0	0-2	1.5	0-5	3.7	0-10
neutrophiles, %	11.3	5-14	12.6	5-29	31.6	6-66
including foamy, %	7.7	2-11	6.0	1-20	16.2	3-34
Erythrocytes:						
mature forms, %	94.1	89.9-96.6	94.1	85.5-99.9	92.6	76.1-99.8
young cells, %	5.9	3.4-10.1	5.9	0.1-14.5	7.4	0.2-23.9

ECOTOXICOLOGICAL ASSESSMENT WATER QUALITY OF VOLGA RIVER

The most common approach to setting environmental regulations, in Russia as well and other country has been based largely on the assessment of chemical attributes of anthropogenic pollution. The system of water quality assessment is based on the concept of Maximum Permissible Concentration (MPC) or Guideline Concentration (GC) of pollutants in the water. At present, the ecotoxicological approach to estimating of water quality gradually meets the approval of more and more researchers. Ecotoxicological assessment of water quality is aimed at obtaining an integrated assessment of water quality, based on symptoms of disturbance in the ecosystem (in situ). The term "ecosystem health" is increasingly used in scientific literature of the past decades. Aquatic ecosystems are stressed in all levels, ranging from individual and up to the population and community levels. For ecosystem health assessment the following four definitions have been used: i) cellular health, which describes the structural integrity of cellular organels and the maintenance of biochemical processes; ii) individual health, which presents structural and morphological health and functioning in terms of physiology of the entire organism; iii) population health, which measures the sustainability and maintenance of a population of a particular species; iv) community health, which describes a group of organisms and the relationships between species in that group. Each method has its limitations and advantages, and the type of method used defines how we interpret the effect of a stressor on ecosystem health (Cairns, 1990; Rapport, 1992, 1995; Calow, 1992; Cash, 1995; Arttril, Depledge, 1997; Elliott et al., 2003). In general, indicators at the biochemical and physiological levels provide information on the functional status of individual organisms, while intermediate-level responses, such as histopathological condition, are indicative of the structural integrity of tissues and organs. Community and population level measurements integrate the responses to a variety of environmental conditions, and therefore may be less reflective of contaminant-induced stress in comparison to the level of organisms (Hinton, Lauren, 1990; Fober, Fober, 1994; Newman, Jagoe, 1996). Many groups of organisms can be used as indicators of environmental and ecological change. But numerous publications attest that fish (in situ) is a good indicator of environmental change and ecosystem health, especially in case of toxic water pollution (Cash, 1995; Wrona, Cash, 1996; Wong, Dixon, 1995; Simon, 2000; Whitfield, Elliott, 2002; Moiseenko, 2005). Fish occupy the top level in the trophic system of aquatic ecosystems. Pathological changes in fish organ enable us to determine the toxicity of water and the potential danger of man-entering substances in water. Fish, in comparison with invertebrate, are more sensitive to many toxicants and are the convenient test-object for indication of ecosystem health. Our results show that water quality and living conditions for aquatic species in the Volga River are unsatisfactory. Based on the prevalence of signs of intoxication in test-organism fish (*Abramis brama* (L.), we can conclude that the ecosystem health conditions are quite dramatic and give a clear signal of the need to decrease toxic pollution. The main question for environmental management is the level to which pollution loading must be reduced to achieve reference conditions and to preserve ecosystem health.

To answer this question, we need to accomplish three tasks (Moisenko *et al.*, 2006):

1. Determinate how hydro-chemical information on water quality can be interpreted in terms of a unified parameter, which could reflect the real impacts of the dose taking into account contaminant complexes (multi-pollution);
2. Assign criteria for ecosystem health that informatively reflect the impacts of pollution;
3. Determine critical levels of water pollution and required load reductions based on a dose–effect relationship.

An integrated impact dose. In rivers and reservoirs, aquatic organisms are exposed to a mixture of all toxicants. It is important to find a numerical parameter describing the total toxic impact on fish. The integrated impact dose of contaminants is determined by their number, concentration, toxic properties of each and aquatic medium – pH, Ca, TOC (Forstner, Wittman, 1983). The values of Maximum Permeation Concentrations (MPC) largely differ by country, in spite of the fact that experimental research techniques to establish the MPCs are universal. In Russia, the MPC values for Cu, V and some other elements are possibly underestimated, whereas the MPCs for Cd, As and some other elements are possibly overestimated. For example, in Canada for Cu and Cd guideline values are 2-4 and 0.01-0.06 μg/l accordingly in dependence of $CaCO_3$, for As it is equal 5 μg/l; in the Netherlands the

MPC value for Cu, V and Cd are 3.8, 5.1 and 2 μg/l accordingly. (Can. Water Qual. Guidelines, 1994; Env. Quality Obj., 2001; Bioassey meth. aquatic org., 1985; Methodological recommendations., 1998). Because the Volga River is in Russia we used data on the toxicological properties of each toxicant based on the MPC adopted in Russia (see Tables 3 and 4). For the investigated areas of the Volga basin Figure 6. presents the total exceedance of the actual concentrations of toxic elements over their respective MPC values. For inorganic compounds, the maximum concentration values, standardized to the respective MPCs, are typical of Mn, V, and Cu. Water contamination with metals (from 13 to 20 units) is typical of the Upper Volga (I, II, III). For the whole set of toxic elements, the most heavily contaminated areas were found in the Gorkovskoe Reservoir (sites IV, V, VI) and the Lower Volga (IX, X, XI, XII, XIII). In the middle and lower courses of the Volga, the toxic properties of water were due to its contamination with organic compounds (mainly with phthalic acid ethers), the sum of the exceedance factors of which reaches 70.

Criteria of ecosystem health. The different types of pathology and dysfunction diagnosed in the bream result from comprehensive chronic impact of numerous toxic substances, found in the Volga River water, on the fish organisms. Histological analysis of fish organs and tissues revealed serious disturbances in the morphology and function of the liver and kidneys, as well as in the hematopoietic system; many of these disturbances are irreversible. These pathologies are based on the physiological reactions through disturbing the homeostasis and proper functioning of vital biological processes.

Determination of the critical levels of water contamination requires numerical biological criteria, which also adequately reflect the effect of toxic substances in the water. Thus, the following biological parameters were used as criteria for fish and ecosystem health (the average weighted for individual river sections):

i) the percentage of fish in which the second or third stages of diseases were diagnosed;
ii) the Z-index defined above;
iii) the percentage of fish with hemoglobin concentration below 90 g/l ;
iv) the low levels of neutrophils in the blood, etc.

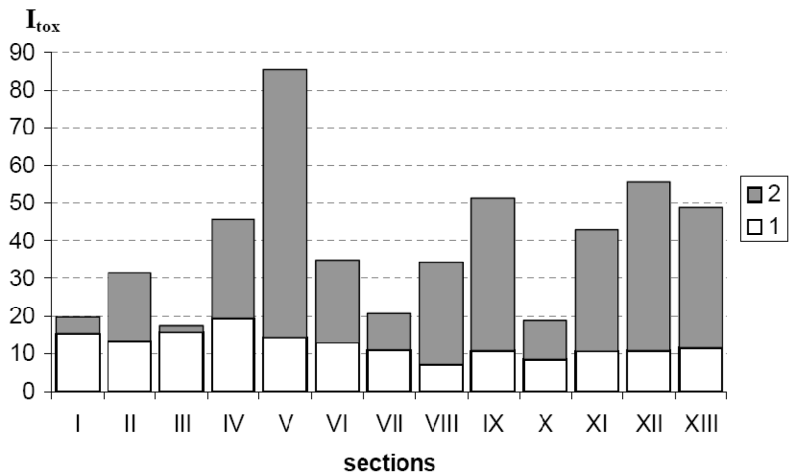

Figure 6. Sum of the concentrations of toxic substances divided by the respective MPC values for the investigated sections of the Volga River (1 - microelements; 2 - organic compounds).

Dose–effect dependencies and critical levels. Basing on dose–effect dependencies (between numerical indices of fish health and the chemical parameters of water quality, in particular the total concentration of toxic substances in the water standardized to MPC), the critical levels of water contamination can be determined. The dose–effect dependencies were plotted for the above biological parameters.

The following factors are assumed to have affected the results: i) the biased nature of the values of MPC, to which the pollutant concentrations were standardized in the process of the integral dose determination (especially for toxic organic compounds); ii) the underestimation of synergetic effects and the presence of other presumably toxic substances in the water, which could also have a negative impact on fish organisms; iii) the persistent effect of toxic substances over the whole lifespan of the fish, the range and concentration of which could be different in different years and seasons; iv) the subjective character of expert evaluation; measurement errors; small samples obtained, etc. However, despite the complexity of the synchronous studies that were carried out and the necessity of accounting for numerous factors, reliable dependencies were obtained. These dependencies confirm that the morbidity in fishes inhabiting the Volga River basin is related to the occurrence of various toxic substances in the water (Figure 7).

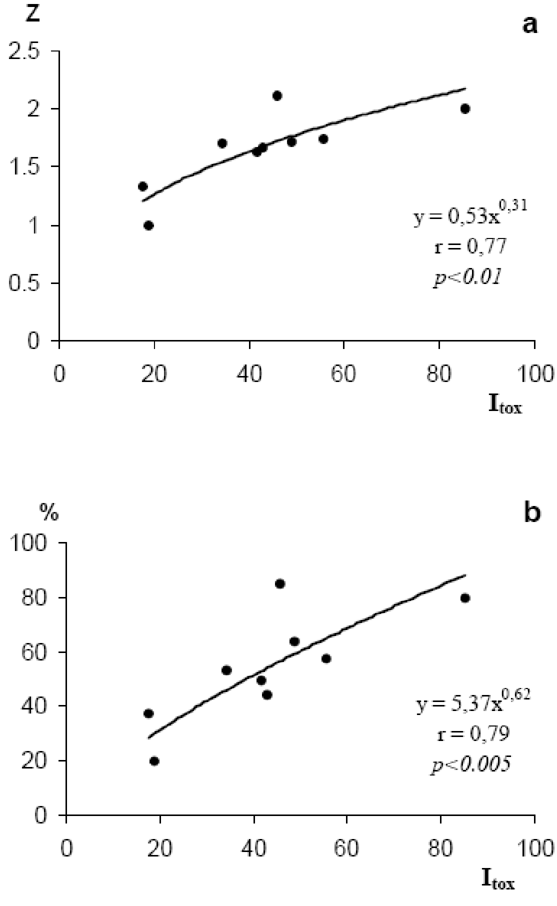

Figure 7. Dependencies of Z (a) and the percentage of fish demonstrating the second and third stages of disease (b) on the total concentration of toxic substances standardized to MPC values.

Table 7. Dependence of characteristics of the physiological state of fish on the concentration of hazardous substances in the water (dash denotes absence of reliable data)

Toxic elements and compounds (x)	Blood characteristics (y)		
	Average Hb	Hb not exceeding 90 g/l	Neutrophiles, %
Dibutyl phthalate	$y = -5.6Ln(x) + 107.2$ r=0.65**	-	-
V	$y = 112.0e^{-0.12x}$ r=0.58*	$y = 52.0Ln(x) + 16.9$ r=0.68***	$y = 6.2e^{0.58x}$ r=0.58*
Pb	$y = 98.2e^{-0.07x}$ r=0.64**	$y = 21.7x + 21.9$ r=0.79****	$y = 10.3x + 9.48$ r=0.88***
	Percentage of the fishes demonstrating different pathologic disturbances in: (y)		
	The whole organism	The liver	The kidneys
Hydrocarbons of oil products	$y = 8.63Ln(x) + 32.4$ r=0.53*	$y = 10.1Ln(x) + 24.6$ r=0.56*	$y = 0.53x + 16.4$ r=0.74***
Dibutyl phthalate	$y = 0.63x + 38.5$ r=0.61*	-	-
Dioctyl phthalate	$y = 0.78x + 45.3$ r=0.59*	$y = 1.17x + 37.9$ r=0.70***	$y = 1.41x + 13.6$ r=0.87****
Derivatives of 1,3-dioxane	$y = 0.47x + 48.7$ r=0.62**	$y = 0.53x + 45.0$ r=0.59*	$y = 0.81x + 20.2$ r=0.87****
Cu	$y = 8.73x + 34.3$ r=0.52*	$y = 10.7x + 25.9$ r=0.57*	-
Mn	-	$y = 24.9Ln(x) - 47.2$ r=0.76***	$y = 0.38x + 5.08$ r=0.68***

*-p<0.05, **-p<0.01, ***-p<0.005, ****-p<0.001

Among the various negative ambient factors that cause pathologic disturbances in fish organs and tissues, it is very difficult to single out the most important factors. Table 7. presents the dependencies between the parameters of fish morbidity and the concentrations of toxic substances in the water. Depletion of certain blood parameters is most significantly related to the impact of V and Pb, whereas pathological disturbances in the fish liver and kidneys are associated with the negative effects of dioctylphthalate, derivatives of dioxane, and oil products, as well as those of Cu and Mn.

The accumulation of toxic metals can also enhance (and, in certain cases, even directly cause) pathologies in fish. Therefore, the relationship between the accumulation of microelements in fish and pathological disturbances in the organs and tissues of bream in the Volga River basin was analyzed. The increase of metals in the water medium may bring adverse effects on fish health. The surplus of trace elements in the organism initiates some specific diseases: Hg causes neurological effects, Cd and Pb have carcinogenic properties, Sr leads to pathology of bone tissues, Cu to anemia, etc. (Conto Cinier *et al.*, 1997; Patriarca *et al.*, 1998; Vatras *et al.*, 1998; Musibono, Day, 1999).

Organisms have mechanisms of metal detoxification by induction of metallothionein synthesis. These proteins bind specifically to neutral essential trace elements, such as Zn and Cu, as well as to potentially toxic metals such as Cd and Hg (Phillips, 1995; Linde *et al.*, 2001). The effects of metal accumulation on fish and their pathologies, without the necessity of explaining the internal metabolism of metals, is the key purpose for our data.

Notwithstanding the low sensitivity of the method applied, which prevented determination of the concentration of Hg in the water, the accumulation of this metal in fish was observed, especially in the Middle Volga. A reliable correlation was established between Hg accumulation in fish kidneys ($Hg_{kidneys}$) and pathologic disturbances in this organ (Pat., %), as well as Z:

$Pat._{kidneys} = 210\ Hg_{kidneys} - 9.68$, r = 0.81, $p<0.005$;
$Z = 53.8\ Hg_{kidneys} + 0.029$, r = 0.85, $p<0.005$.

Thus, irrespective of the fact that the concentration of Hg in the investigated water was lower than the analytical detection limit (less than 0.05 μg/l), its accumulation in the organism can cause pathogenic disturbances in fish.

A reliable correlation was also established between the accumulation of Cd in fish gills (Cd_{gills}) and hematologic parameters of fish –such as the concentration of hemoglobin in the blood (Hb), and neutrophils (N) in the leukocyte count.

$Hb = -104\ Cd_{gills} + 103$, r = 0.87, $p<0.001$;
$N = 34.5\ Cd_{gills} + 3.35$, r = 0.88, $p<0.001$.

As mentioned above, the accumulation of Cd in the organism is accompanied by an increase in the concentrations of some other elements. Most probably, the joint accumulation of several toxic elements in the fish organism entails a decrease in the concentration of hemoglobin in the blood and the development of anemia, accompanied by an increased percentage of neutrophils in the leukocyte count. In addition, a correlation between the concentration of Pb in fish kidneys (Pb_{kidhey}) and pathological disturbances in this organ (Pat_{kidney}) was established.

$Pat._{kidneys} = 53.2\ Pb_{kidneys} + 17.2$, r = 0.54, $p<0.1$.

All this testifies to the fact that accumulation of metals (especially Hg and Cd) leads to pathological conditions in fish. Thus, the increase in the metal concentrations in the Volga River basin results in their accumulation in fishes, leading to the development of microelementoses and pathologic disturbances in fish organs and tissues.

The established critical levels of water contamination remain open for discussion. The studies carried out by the authors have shown that the water quality and ecosystem health in all the investigated river sections are unsatisfactory, and that critical levels of water contamination are exceeded. Approximation of the dependencies into the area of low values of the water quality standard (less than 1 unit) shows that the percentage of fish in which the second or third stage of disease was diagnosed was equal to about 10% (Figure 7).

The dose–effect dependencies clearly show that total pollution of the Volga River must be significantly decreased, by at least 5–7 times, first for toxic contaminants. These studies have confirmed the high information value of the ecotoxicological approach to the assessment of water quality and ecosystem health. Note that ecotoxicological studies were carried out for the Volga River basin for the first time, and many important river sections or reservoir areas were not investigated. In this respect, our studies could be considered "screening analysis of the ecotoxicological situation," but at the same time, they substantiate the information content of methodological solutions and the necessity of the continuation of large-scale studies in this field in the future.

CONCLUSION

The Volga is the longest river of Europe. Large-scale contamination of the Volga River basin is caused by its geographical position within the most economically developed region of Russia. Domestic and industrial wastewaters, air-borne pollution of the catchment area, as well as non-sewerage effluents from settlement areas find their way to this water basin.

Numerous elements and their compounds that have a toxic effect on living organisms were found in the water samples taken within the investigated sections of the Volga River. Among inorganic substances, V, Cu, and Mn play the most important role in the formation of the general ecotoxicological situation. As for organic compounds, a high level of water contamination with phthalic acid ethers and dioxane derivatives was first recorded. In the investigated sections of the Upper Volga, water contamination with metals prevails; in the Middle and Lower Volga, contamination with organic xenobiotics prevails. The highest levels of the total exceedance of the actual substances concentration over the respective MPC values were observed for the Gorkovskoe Reservoir and certain sections of the Lower Volga. Morphological and functional disturbances in the organs and tissues of fishes testify to their intoxication. Most of the fishes with different forms of pathology and dysfunction were caught in the Gorkii Reservoir and in certain sections of the Lower Volga (downstream of Astrakhan).

Results of the research testify to the fact that the examined fish individuals are subject to the effect of multicomponent “chronic” water contamination. Numerous registered disturbances (necroses, neoplasms) are referred to as irreversible. However, hypertrophy, hyperplasia, and encapsulation, accompanying the above disturbances, are structural and functional bases of adaptive reactions aimed at surviving of fish under the conditions of subtoxic aquatic environment.

Hematologic characteristics of the examined fishes confirm the fact of their intoxication. Symptoms of anemia and increased concentration of neutrophiles and monocytes were found. All this is the response of the organism to unfavorable habitat conditions. On certain blood smears, numerous pathological forms of blood cells (laky erythrocytes, poikilocythemia, vacuolization of the cytoplasm, pycnosis of the cell nuclei, amitosis of the cell nuclei, etc.) were found. They testify to disturbances in the system of hemogenesis of fish caused by toxic substances.

Based on the dose–effect dependences, it has been found that diseases of fish are caused by water contamination with toxic substances. The negative impact of organic xenobiotics on the fish liver and kidneys has been demonstrated, in addition to the negative impact of certain microelements (e.g. vanadium, lead and some other ones) on the hemogenesis system. The studies that were carried out confirm the high information value of the ecotoxicological approach to the assessment of the ecological state of water bodies, as well as the necessity of establishing more reliable MPC values and maximum permissible "Toxicological Harmfulness Value". Note that ecotoxicological studies were carried out for the Volga River basin for the first time, and many important river sections or reservoir areas were not investigated. In this respect, our studies can rather be called "screening analysis of the ecotoxicological situation," but at the same time, they are convincing for substantiating the information content of methodological solutions and the necessity of continuation of large-scale studies in this field in the future.

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Chapter 6

ECOHYDROLOGY-BASED PLANNING AS A SOLUTION TO ADDRESS AN EMERGING WATER CRISIS IN THE SERENGETI ECOSYSTEM AND LAKE VICTORIA

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1. INTRODUCTION

The Serengeti ecosystem is often taken to be the 25000 km^2 animal migration area (Figure 1a). This includes the 14,763 km^2 Serengeti National Park (SNP), the Masai Mara Reserve in Kenya, and a number of game controlled areas that form a buffer zones, principally the Maswa, Ngorongoro, Loliondo, Ikorongo, Grumeti, and the Speke Gulf Game Controlled Area (SGGCA) that, although tiny (95 km^2), is potentially important because, if human encroachment was removed, it would provide access for wildlife to the permanent waters of Lake Victoria (Figure 1b). However this definition of the ecosystem ignores the hydrology. The Serengeti ecosystem has only one perennial river, the Mara River. The Mara River, together with a few scattered springs in the northern region of the SNP, is the only source of water for migrating wildlife in the dry season in a drought year. Thus the source of Mara River water in the dry season, namely the Mau forest in Kenya's highlands, is also part

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of the Serengeti ecosystem even if the migrating animals do not migrate to that area (Gereta et al., 2002 and 2009).

Lake Victoria (Figure 1c) has a surface area of 96,000 km^2. Its shores are shared by Uganda, Kenya, and Tanzania. It drains 25 large rivers with a total catchment area of about 184,000 km^2, supporting 30 million people. The lake is large enough (see below) to affect rainfall over the Serengeti (Figure 2).

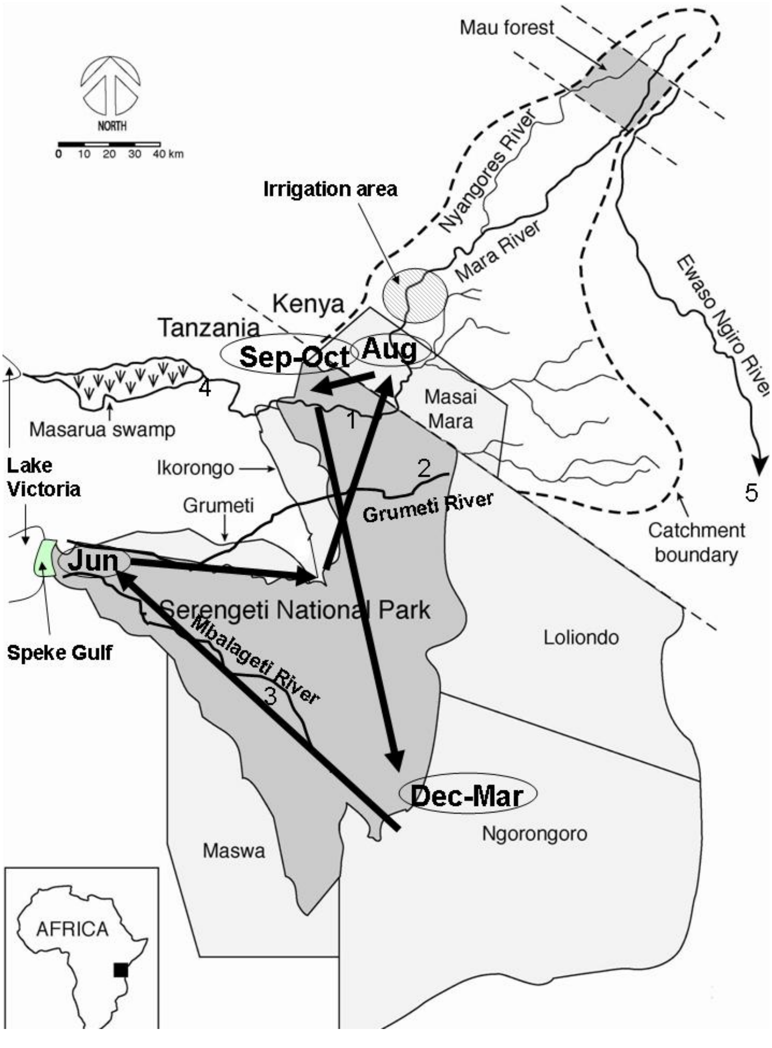


Figure 1a. Map of the Serengeti ecosystem showing the Serengeti National Park (SNP) and the buffer zones including hunting and game reserves and managed conservation areas in Kenya and Tanzania. The Mara River is the main source of drinking water for the migrating animals in the dry season in a drought year. The future of this water is threatened by deforestation of the Mau forest and by irrigation farming in Kenya. The Mau forest is also the source of water for the Ewaso Ngiro River which flows east of the Serengeti ecosystem. Water monitoring sites: 1 = Mara River at Kogatende; 2 = Grumeti River at road crossing; 3 = Mbalageti River at Lake Magadi; 4= Mara River at Mara Mine; 5 = Ewaso Ngiro River at Ketri bridge near (another) Lake Magadi. The arrows and months show the wildebeest migration in the Serengeti-Mara ecosystem during 1999–2000 (simplified from Thirgood et al., 2004).

Figure 1b. The proposed extension of SNP to Lake Victoria covers the Speke Gulf Game Controlled Area (SGGCA) and includes about 15 km^2 of papyrus wetlands located mainly at the mouth of rivers, and a strip of inshore lake waters (Source: Google image, 2008).

Figure 1c. Map of the drainage area of Lake Victoria, showing the location and approximate boundaries of the Serengeti National Park (SNP) and Rubondo Island National Park (RNP).

The Serengeti ecosystem is drained by the Mara, Grumeti and Mbalageti rivers that all flow westward into Lake Victoria. The Mara River passes through SNP in the far north, at 1,300 to 1,500 meters of elevation, and drains a total of 10,300 km², most of which is in Kenya. The Grumeti River, with a catchment area of 11,600 km², drains much of the wooded savannah of the central and northern hills, much of which is inside the park's boundaries. To the south, in an area of treeless grasslands and hills lying between 1,600 and 1,660 meters in elevation, the Mbalageti River drains 2,680 km², nearly all within the park. River flows were gauged (Brown et al., 1981) and found to vary greatly with the seasons. After the rains, discharges decrease exponentially to base flows maintained by ground water seepage. In the 1970s the Mara River reached base flow four months after the rains end, whereas as a result of deforestation and irrigation in Kenya (Figure 1) base flow in 2004-2005 was reached after only one month. The Grumeti and Mbalageti rivers typically decline to essentially zero flow within a few weeks. Thus during much of the dry season, the only water remaining in the Grumeti and Mbalageti rivers is stagnant pools tens to hundreds of meters long and usually less than a meter deep, siltation being reduced by stirring by hippos. Whether there is exchange of water, or not, between water holes through groundwater is unknown.

2. Rainfall Land Lake Victoria Water Level

Serengeti rainfall data were collected on a monthly basis from January 1960 to December 2006 from 232 stations, typically 10% being operational at any one time during that period (Wolanski and Gereta, 2001).

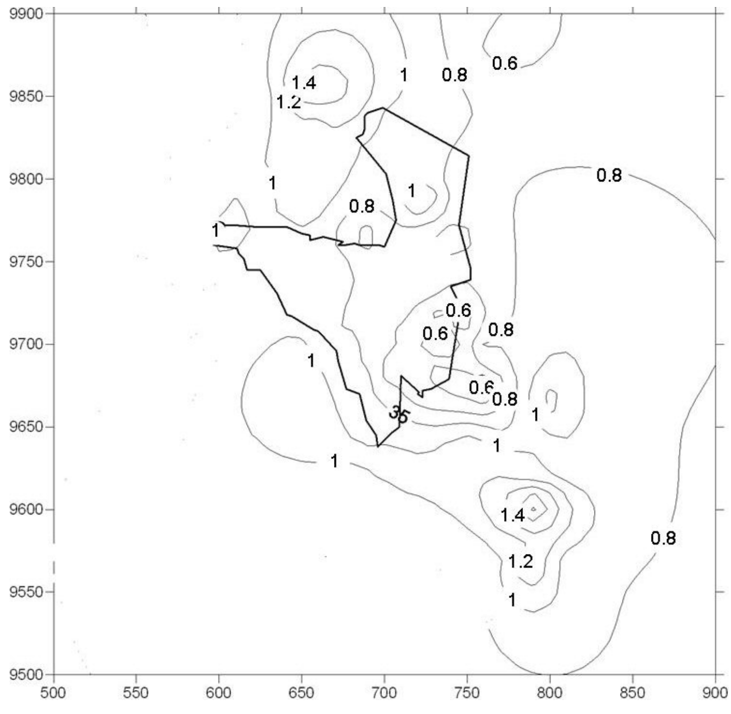


Figure 2. Relative mean annual rainfall distribution.

The mean annual rainfall over the Serengeti was not distributed uniformly (Figure 2); it was greatest in the northern and western corners of the SNP, in the Ngorongoro, and least in the southern grasslands, which are in the rain shadow of the Ngorongoro mountains. The presence of Lake Victoria with its higher rainfall explains the higher rainfall over the westernmost region of SNP and surrounding areas.

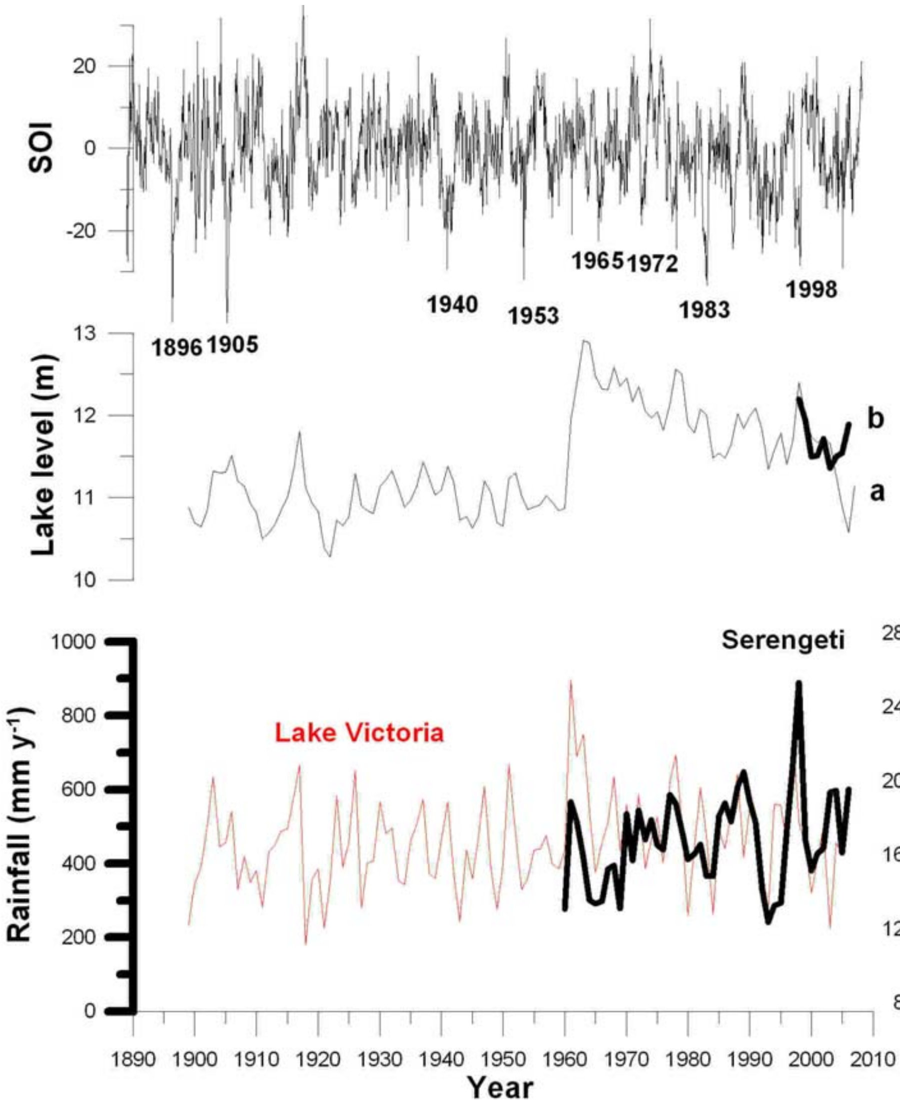


Figure 3. Time-series plot for 1899-2007 of the Southern Oscillation Index (SOI), Lake Victoria water level as (a) observed (a) and (b) what it should have been if Uganda did not overdraw water since the year 2000 (Kiwango and Wolanski, 2008), and the mean annual rainfall over Lake Victoria and the Serengeti.

Mean rainfall over the SNP in the period 1960-2006 was 468 ± 127 mm y^{-1} with a large interannual variability (± 127 mm y^{-1}) that was not correlated with the Southern Oscillation Index (S.O.I., which is a measure of the El Nino – La Nina phenomenon; Figure 3; Wolanski et al., 1999; Wolanski and Gereta, 2001). However wet years (highest 10% rainfall) and extreme drought years (lowest 10% rainfall) generally, but not always, occurred during El Nino and La Nina events, respectively. The interannual variability of rainfall is thus not attributable uniquely to the El Nino – La Nina phenomenon, and depends on a number of atmospheric teleconnections including the Quasi-Biennal Oscillation (McIntyre, 1993; Jury et al., 1994; Jury, 1996).

Rainfall over the Serengeti fluctuated seasonally (Table 1). Rainfall was bimodal, with the major peak in April and a secondary peak in December. There was a large interannual variation, as evidenced by the maximum monthly rainfall being several times larger than the mean monthly rainfall and by the minimum monthly rainfall being several times smaller than the mean monthly rainfall.

Table 1. Mean, maximum, and minimum rainfall (mm month^{-1}) over the Serengeti. The data were spatially averaged and cover the period 1960-2006

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Mean	50.5	48.3	66.0	75.3	38.9	14.5	8.9	9.7	14.2	21.2	48.2	63.0
Max	192.3	170.0	149.3	153.8	107.1	49.7	69.9	28.4	50.6	82.0	117.5	266.0
Min	14.2	16.8	15.2	16.2	9.3	0.0	0.7	2.0	0.0	3.5	9.0	6.7

The mean rainfall over Lake Victoria in the period 1960-2006, calculated using Nicholson and Yin (2001) model based on the history of elevation of lake Victoria (Figure 3), was 1749 ± 270 mm y^{-1}, which is much higher than that over the Serengeti during the same period. Rainfall over the Serengeti is weakly correlated with that over Lake Victoria (R^2=0.5; p>0.05). Mean rainfall over lake Victoria during the period 1899-1959 was smaller (1610 ± 220 mm y^{-1}), suggesting increasing rainfall over the last 46 years. Rainfall over Lake Victoria also showed large interannual variability (Figure 3), which is not correlated with the S.O.I. (R^2 = 0.3). The mean lake rainfall contributes 82% of the total inflow; the remaining 18% is contributed from river inflow. Of the outflow, 76% is due to evaporation from the lake and the remaining 24% is the natural outflow forming the White Nile River at Jinja (Scheren et al., 2000; COWI, 2002).

The water level of Lake Victoria has fluctuated in the last century, but no decrease has been as drastic as that from 2000 to 2006 (Figure 3). The water level shows a discontinuity in 1961, though the SOI for that year does not show a large El Nino phenomenon. The lake level started dropping anomalously in 2000 (lines a and b in Figure 3) and in October 2006, the lowest lake level in 83 years was recorded, just 16 mm above that of 1923. In 1959, the Owen Falls dam (now known as the Nalubaale dam) was constructed for hydroelectricity generation, transforming Lake Victoria into a huge reservoir. The operation of this dam was based on an "Agreed Curve" (based on agreements in 1949, 1953 and again in 1991 between Uganda and Egypt), whereby the dam outflow was to mimic the natural flow (Kull, 2006). Thus the lake level fluctuated in time but remained that determined by natural conditions. In 2000 the new Kiira dam was completed at Jinja, also for hydroelectricity generation. Because the Kiira and Nalubaale dams operate in parallel, more water is needed to generate hydroelectricity at the two dams at the same time and thus it became impossible for Uganda to adhere to the ‘‘Agreed Curve’’; the White Nile River discharge was increased by 30% to 50% (Figure 4), and the lake level dropped as a result (Awange et al., 2007; Kiwango and Wolanski, 2008).

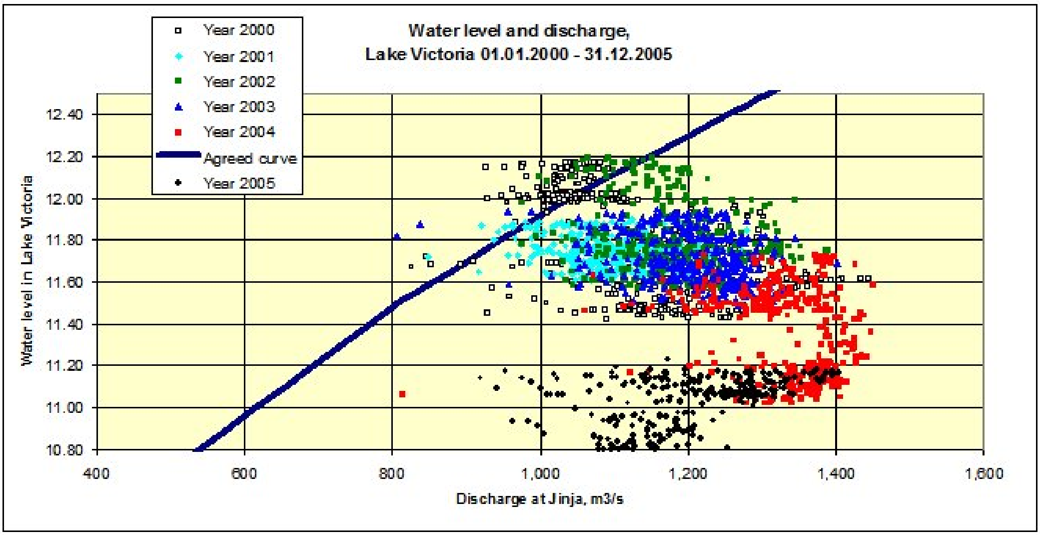


Figure 4. Scatter plot of the Lake Victoria water levels versus the measured outflow from Lake Victoria at Jinja that forms the Nile River discharge in 2000-2005, demonstrating the overdrawing of water above the allowable discharge as set by the ’Agreed curve’. (Source: DWD, Uganda).

3. Rainfall and Vegetation

These rainfall patterns are clearly reflected in vegetation, with the boundary between wooded savannah and grassland moving southward during the wetter decades (Gereta et al., 2004). Nonetheless, other factors may influence this boundary shift, including a change in man-made fires and the discouragement of poaching of elephants (Gereta et al., 2004).

4. Rainfall, Salinity, and Animal Migration

The interannual variability of rainfall controlled the interannual spatial and temporal variability of the migration of large ungulates (Gereta and Wolanski, 1998; Wolanski et al., 1999). The spatial variability is directly determined by the rainfall, the wet season dispersion area of the migrating herds of both zebra and wildebeest in the southern grassland increases (decreases) with larger (smaller) rainfall (Figure 5). In turn the size of the dispersion area determines the available nutrition (Mduma et al., 1998), mineral nutrition being a key reason why the animals migrate in the SNP (McNaughton, 1979, 1985, 1988, 1990; McNaughton et al., 1998). The ponded surface waters in the southern grasslands become highly saline at the end of the wet season before drying out during the dry season (Figure 6). The timing of this high salinity is controlled by rainfall during the wet season. The onset of the northward migration away from the southern grassland into the wooded savannah at the start of the dry season is controlled by the high salinity because the animals leave the grassland when the water becomes too saline (salinity > 5 psu) to be drinkable by mammals, and this happens even if there is still fodder available. The timing of this migration can be predicted up to one month in advance in a dry year and three months in advance in a wet year, with an accuracy of a few days, from a rainfall-evaporation-salinity model (Wolanski and Gereta, 2001).

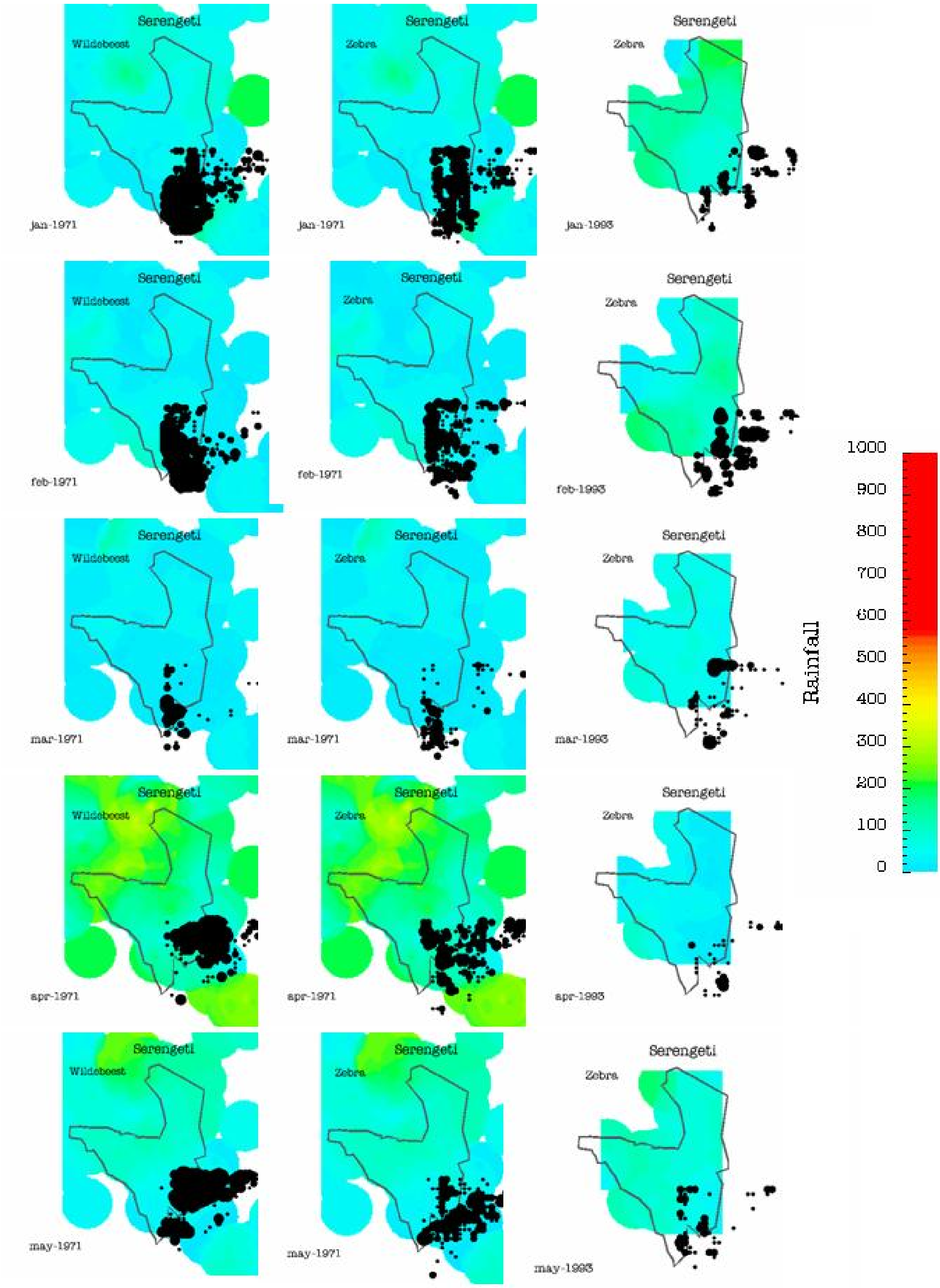


Figure 5. Spatial distribution of the wildebeest and zebra during the 1971 wet season, and of the wildebeest during the 1993 wet season. 1971 and 1993 were respectively a wet year and a dry year. Rainfall in 0.1 mm month^{-1}. Note that the animals migrated southward during the wet season in greater numbers during a wet year than a dry year.

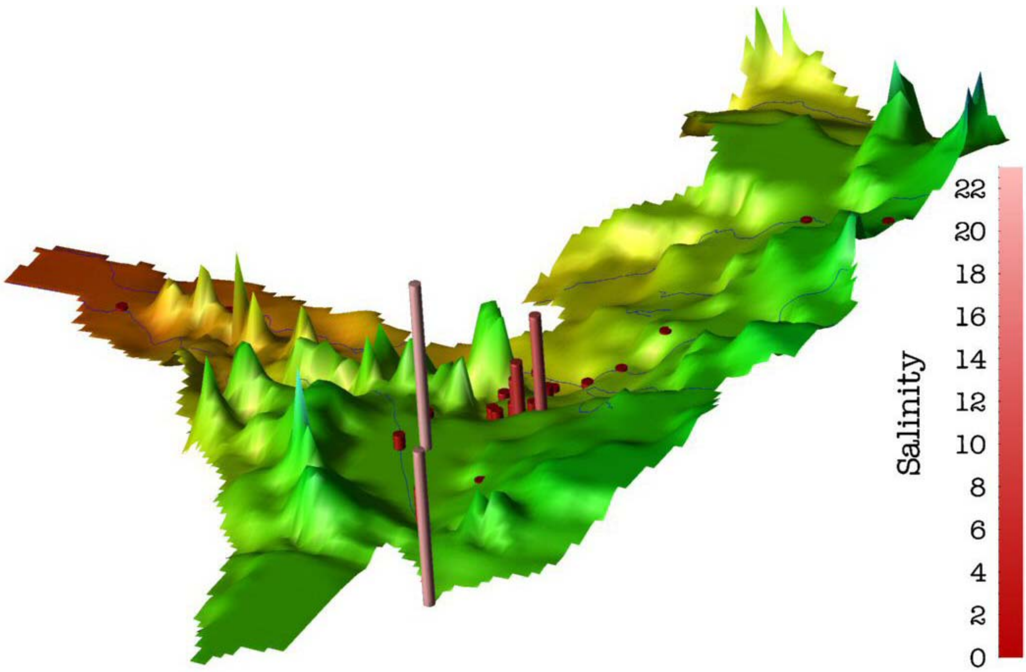


Figure 6. Salinity (psu) of surface waters in the SNP in April 1996, near the end of the wet season.

5. WATER QUALITY IN PONDED WATERS

The water quality of the stagnant water pools in the dry season is very poor as the waters are highly eutrophicated by animal dung. Algae and suspended solids (mainly organic detritus) restrict the photic zone to the top 2 cm of the water column, and this creates a strong temperature stratification (~ 2 ^{o}C m^{-1}). The dissolved oxygen concentration (DO) fluctuated widely (by up to 11.5 mg l^{-1}) as a function of time, mechanical stirring and aeration by animals, mainly hippos, and the presence of fringing wetlands (Wolanski and Gereta, 1999; Gereta et al., 2004). The DO cycle can be modeled (within an accuracy of 0.3 mg l^{-1}) by assuming (Figure 7) that the four dominant processes were photosynthesis and respiration by algae near the surface, trapping by wetlands, decomposition of dead organic matter on the bottom, and stirring/aeration by hippos (Mnaya et al., 2006). The rate of DO decline from the decay of dead organic matter was equal to the rate of DO removal by algal respiration at night. Anoxia (DO=0) and hypoxia (DO < 2 mg l^{-1}) occur commonly, especially at night and near the bottom. Since every water pool differs in size, in the extent of fringing wetlands, animal dung, and resident hippo population, the diurnal DO cycle is different for each pool, and this in turn impacts the utilization by wildlife of individual pools; this introduces patchiness in the biodiversity.

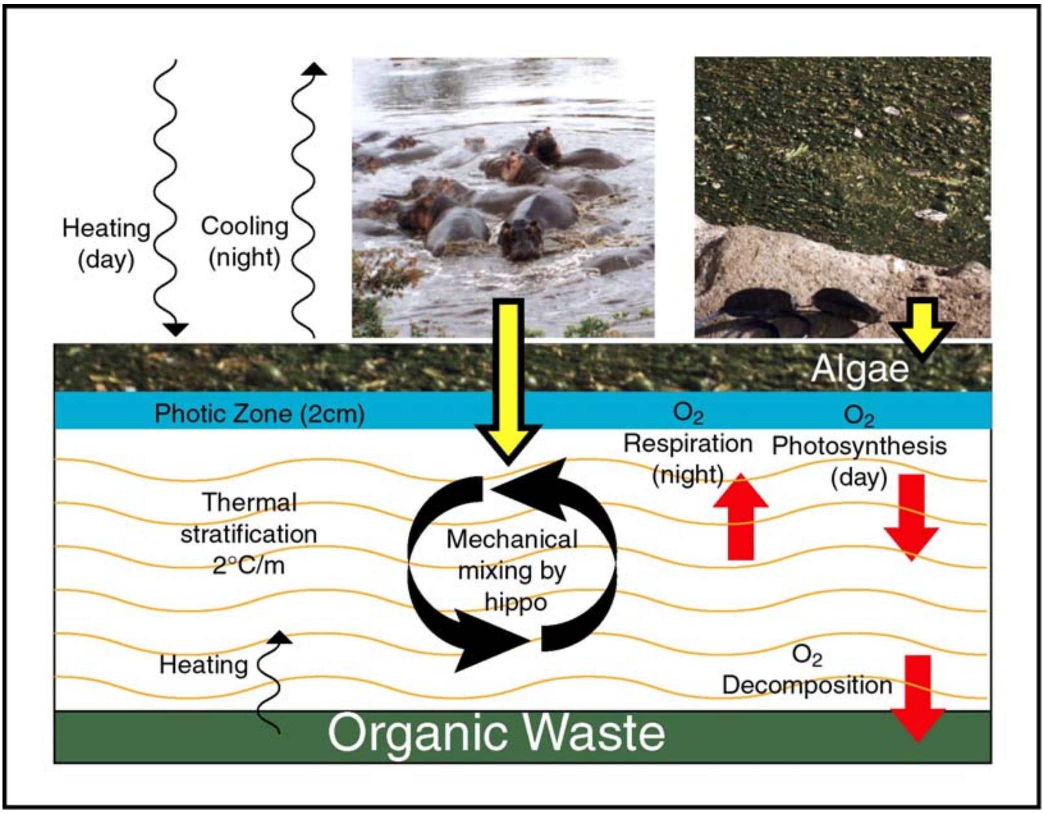


Figure 7. Sketch of the processes controlling hypoxia in dry season, stagnant water ponds in the SNP.

6. Water Quality and Fisheries

Because of poor water quality in the dry season the SNP surface waters are thus hospitable to only the hardiest species of aquatic flora and fauna, with the exception of the Mara River, which is flowing and does not suffer anoxia or hypoxia. The biodiversity of flora and fauna in these waters remains largely unexplored. Recent discoveries by Farm (2000) revealed a new species of fish, *Barbus serengetiensis* (Teleostei: Cyprinidae). The fish was found in shallow streams and rivers with gravel or sand substrata. The specimens were collected in streams where visibility was less than 10 cm. At present the distribution of this new species is only known from the Orangi River (a tributary of the Grumeti River) and some of its tributaries: the Pololeti, Seronera, Ngarenanyuki and Nyabogati Rivers, all of which are located in the centre of SNP. The Mara River, being perennial, supports a much larger fish population. The current level of fish stock in the Mara River is not established, but research work by TAFIRI shows that most of the fish species that have disappeared from Lake Victoria are now found in Mara River. Such genera include *Oreochromis* and *Esculenta*. Some rare species like *ctenopoma murici* are also found in this river. Apart from being a good habitat for rare species, the river is an important breeding ground for fish.

Outside the SNP, the fishing activities are sometimes intense but only at subsistence level. People use local fishing gear to catch fish for food and to earn some money. Average income from fishing is about TAS 10,000 per head per day. If the Mara River dries (see below), the fishing community will be denied of this income. Additionally, the inhabitants of this area will be denied of the food that is very rich in nutrients. And above all, the potential for improved fish production is immense.

8. Importance of Papyrus to Prevent Eutrophication of Lake Victoria, Provide Food Security, and Decrease Poaching in the SNP

The watershed of Lake Victoria includes vast wetlands, both within the catchments and along the shores of the lake, most of which are composed of monotypic stands of papyrus plant covering about 10235 km^2 (Kansiime and Nalubega, 1999; Kiwango, 2007). Papyrus is a large herbaceous sedge species with a culm growing up to a height of 5 m (Jones and Muthuri, 1985). Fringing papyrus wetlands are an important buffer zone protecting the lake from eutrophication. Growing papyrus absorb organic nutrients from both water and sediment and thus trap nutrients especially nitrogen and phosphorus (Gaudet, 1979; Kassenga, 1997; Kansiime and Nalubega, 1999; Azza et al., 2000; Kyambadde et al., 2004; Gichuki et al., 2005). During senescence, the papyrus plants accumulate nutrients in their root zones and the decaying papyrus release nutrients back in the water (Asaeda et al., 2002). About one third of the dead biomass is deposited back in the wetland; the rest is lost to elution, rain and decomposition (Gaudet, 1977; Muthuri and Jones, 1997). Denitrification is the only process in the nitrogen cycle that permanently removes nutrients (Kansiime and Nalubega, 1999).

Lake Victorian papyrus wetlands can remove up to 53% annual nitrogen input by denitrification alone, while potentially, above-ground biomass harvest can remove 26% and 19% annual nitrogen and phosphorus input per year respectively (Kiwango and Wolanski, 2008).

Papyrus wetlands are also important to the ecology of the tilapia fish, by providing a refuge for the juveniles (Mnaya and Wolanski, 2002). Although the commercial fish in the lake is the Nile Perch, this fish does not depend on the papyrus wetlands as does the tilapia. The tilapia supports the artisanal fisheries. The decrease of the lake level (2000 – 2006) lowered the water level below that of the intake at Kiira dam and stopped the water extraction. Unseasonal large rainfalls occurred in November–December 2006 and in January–February 2007. The lake level rose by 1 m in the first six months of 2007 and inundated the papyrus wetlands again (Figure. 3). Studies observed that the papyrus at Mlaga Bay had recovered fully by September 2007. The papyrus survived 18 months of exposure because the peat that the rhizome and roots grew into retained enough moisture at 0.5–1 m depth for the plant to remain alive. If un-seasonal large rainfall did not occur, as it did, the lake level would not have risen and over time the peat in the papyrus wetlands would have dried out and the papyrus died. The above-ground papyrus biomass would either be burned for land clearance for cultivation, access to the lake, settlements, and removal of pest animals (Awange and Ong'ang'a, 2006), or would have decayed and ultimately ended up in the lake. Kiwango and Wolanski (2008) showed that if the wetlands in the lake basin did dry and the dead plant matter did end up in the lake, it would be equivalent to 18,500 years of nutrient inputs to the lake, greatly accelerating eutrophication. Alternatively, if the papyrus was burned, $4.1 * 10^5$

tonnes, (i.e. 17% of the annual input of N into the lake, thus also increasing the lake eutrophication) would end up into the lake, and 5.2 * 10^7 tonnes of CO_2 would be released to the atmosphere, which is about 5% of the CO_2 released by the catastrophic peat and forest fires in Indonesia during 1997 (Pege et al., 2002).

During December 2000 and December 2006, fish larvae were sampled each night at the same location in front of papyrus wetlands at Mlaga Bay, Rubondo Island National Park (RNP) following the method of Mnaya and Wolanski (2002). The substrate of the papyrus wetlands at Mlaga Bay was underwater in 2000; it was exposed in 2006 when it thus offered no refuge to the fish. The fish larvae recruitment at that site decreased by 80% from 2000 (10.6 fish per sample ± 13.4) to 2006 (2.4 ± 1.5). This decrease is statistically significant ($t(23) = 2.967$, $p < 0.05$).

In 2005–2006 the newly exposed shorelines of Lake Victoria were quickly encroached for settlements, grazing, and cultivation and the exposed papyrus wetlands were replaced by agricultural crops (Awange and Ong'ang'a, 2006). Thus the cconsequences of water level decrease due to overdrawing at water by Uganda include exposing shores which are encroached for settlement, grazing and cultivation, a loss of the buffering offered by the papyrus, and the threat to the tilapia artisanal fisheries. The impoverished people living along the lake's shore depend on tilapia for their livelihood and food security; these are the people who do not have access to the Nile Perch (Odada *et al*., 2004) and whose food security is compromised by drying of wetlands by the overdrawing of water for hydroelectricity in Uganda between 2000 and 2006. This threat to the food security may also threaten the Serengeti because it may lead to increased poaching in the SNP as people look for protein, in a similar manner that the collapse of the artisanal coastal fisheries in Guinea in the late 1990s by allowing commercial trawlers over the nearshore artisanal fishing grounds, lead to the empoverished, artisanal fishermen switching to poaching for bushmeat.

The long-term survival of the Serengeti may thus depend on maintaining tilapia artisanal fisheries and the papyrus wetlands along the shores of Lake Victoria. Maintaining the fisheries requires preventing eutrophication, and this may be facilitated by sustainably harvesting papyrus above-ground biomass for the purpose of nutrient removal (Kansiime and Nalubega, 1999; Kiwango and Wolanski, 2008). This harvesting of papyrus plants for the purpose of nutrient removal requires careful timing. This is because depending on the age of the plants, only 5–20% of the total nutrients may be stored in harvestable parts of the plants (Wetzel, 1975). About 5.69 * 10^4 kg N year-1 and 9.38 * 10^3 kg P year-1 can be removed by harvesting the above ground biomass in a 0.92 km^2 papyrus wetland area (Kansiime and Nalubega, 1999). The above-ground biomass of papyrus is 4.77 * 103 tonnes km^{-2}, while its nutrients content is 1.30% for N and 0.21% for P. Thus if all the above-ground papyrus of Lake Victoria was harvested in one year, this would remove 6.3 * 105 tonnes N $year^{-1}$, or about 30% of the annual load of N to the lake. For P, the average nutrient content in aboveground biomass is 1.02 * 105 tonnes P year-1. About 5.34 * 105 tonnes P $year^{-1}$ enter the lake (COWI, 2002). The amount of P that could be removed by harvesting all the above-ground biomass of papyrus wetlands is thus equal to 19% of the annual input of P. If there was no more input of P, the time scale required for removing the P from the lake if all papyrus wetlands were harvested is 2 years. Therefore, regular harvesting of the above-ground biomass of papyrus plants and removing this organic matter for use in the watershed may be a practical alternative for removing nutrients from the lake, thus reducing Lake Victoria's eutrophication. It would extend to the scale of Lake Victoria the practice of harvesting emergent plants from ponds in urban settings to remove nutrients from the water (Zalewski, 2002).

9. The Key Role of the Mau Forests to Maintain the Serengeti Migration

The Mau Forest is a high rainfall area (1400 mm y^{-1}) of high elevation (3000 m asl.) on the steep slopes of the Mau escarpment which forms part of the western slope of the Rift Valley in the western highlands of Kenya (Figure 1). It is one of the few remaining moist forests in Kenya. This forest regulates water run-off, maintains perennial flow in rivers, prevents soil erosion, limits downstream siltation, and prevents flooding (Dwasi, 2002; Gereta et al., 2002 and 2009). The need to preserve the Mau Forests was noted by the colonial government, hence, its gazettment as early as 1902 specifically for the protection and conservation of water catchment areas. Besides supporting an abundant wildlife, by providing water in the dry season the Mau forest also supports the flamingos in Lake Nakuru and the lesser flamingos in Lake Natron. Though gazetted, the forest is increasingly destroyed by encroachment, both legal and illegal. For example during the 1970s the Mau forest covered about 50% of the Lake Nakuru catchment basin, at present only about 10% remains forested.

The Mara River catchment is a trans-boundary catchment with 65% of it in Kenya and the remaining in Tanzania. The forested area of the Mara River in the Mau escarpment covered 752 km^2 in 1973 and only 493 km^2 in 2000 and deforestation is continuing to 2010 though at a decreased rate (Figure 8).

On leaving the Mau forest, the Mara River passes through an irrigation area that extracts up to 75% of its dry season flow, it then enters the Masai Mara Reserve and then it turns southwestward and crosses the SNP before entering Masarua swamp and finally draining into Lake Victoria. The lower portion of the Mara River in Tanzania is a dry plain of lower elevation of about 1300 m asl. receiving between 700 mm and 1200 mm of rainfall per year with high evapotranspiration and considerable water loss. The Mara River has been gauged daily in 1970-1974 and in 2004-2006.

The Ewaso Ngiro River drains a similar area of the Mau forest as the Mara River. It flows initially nearly parallel to the Mara River before turning southeastward toward Lake Natron. It was gauged in 1948-1990.

Hydrologic studies (Gereta et al., 2009) found that the dry season flow in the Mara River increased by 20% from 1960 to 1990, possibly due to an increase in rainfall (see above). In 2005 however the dry season flow of the Mara River was 68% smaller than the dry season flows in the 1970s. This can be attributed to deforestation of the Mau forest and irrigation just upstream of the Masai Mara Reserve (Figure 1). Hydrology models suggest that if the 1948 and 1972 droughts re-occurred (when the Mara River never stopped flowing) the Mara River in SNP would dry out for 2 months and 1 month, respectively (Gereta et al., 2009).

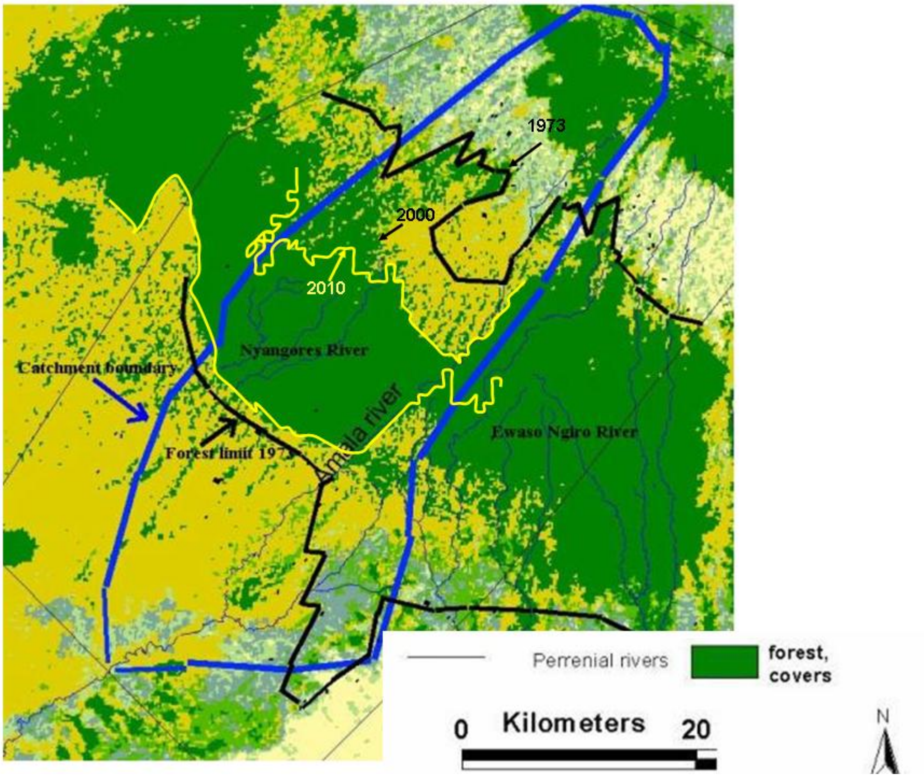


Figure 8. Landsat photograph of January 12, 2000, of the Mau forest in the Mara River catchment, together with the forest limit in 1973 (black kine; source: Landsat) and in 2010 (yellow line; source: Google image). Note that the rate of deforestation has decreased in recent years but has not stopped.

To study the evolution of the Serengeti ecosystem under various scenarios of climate change and human-induced drying out of the Mara River in drought, the ecohydrology model of Gereta and Wolanski (2002) was used. The ecohydrology model has three trophic layers. The bottom trophic layer is the grass, which grows when watered and withers in the absence of rainfall. The grass is grazed by herbivores. The herbivores calve once a year. The herbivores can die from poaching (for which data are available from the Park Warden), starvation (in the dry seasons) and disease (mainly in the dry season; Mduma et al*.,* 1999). The carnivores prey upon the herbivores. The model ecosystem is divided in two areas. Area A (the southern grasslands) is used by the herbivores in the wet season. Area B (the northern region along the Mara River in the Park and in the Masai-Mara National Reserve) is the dry season refuge for the herbivores. At the start of the dry season the herbivores migrate from area A to area B; they return to area A at the start of the wet season. The migration results in an additional mortality of the herbivores. The model bulks together all herbivores and all carnivores. It is necessary for model calibration to scale the herbivore population by that of the wildebeest, and the carnivore population by that of the lions, because data are available on the number of wildebeest and lions. It is thus assumed that the wildebeest form a constant proportion of the herbivore population, and the lions a constant proportion of the carnivore population. In this way, other carnivores, such as hyenas, are also implicitly included. The availability of grass for food for the migrating ungulates was controlled by rainfall. The water availability was controlled by the hydrologic model of Brown et al. (1981), and only kicks in during a drought to estimate if the animals have insufficient water for drinking, this did not happen in historical conditions 1960-2000 for which data are available. The model equations are of the Lotka-Volterra type for biomass at each trophic level, they express mass conservation (see Gereta et al., 2002 and 2009; Gereta and Wolanski, 2008).

The model reproduces successfully (line a in Figure 9) the observations of Packer and Gereta (unpubl. data), Wolanski et al. (1999), and Serneels and Lambin (2001), on the number of wildebeest (and lions; not shown) between 1960 and 1999. The model was also successful in reproducing the 20% die-off of wildebeest during the 1993 drought (Mduma et al., 1999). Predation by lions is predicted to be of secondary importance for the herbivore population dynamics in the Serengeti ecosystem. The model reproduces Mduma et al. (1999) observations that the population of herbivores is limited by the availability of water and forage, and thus fluctuates inter-annually as a result of rainfall. It is apparent that the population of wildebeest did not reach a quasi-steady state (i.e. a population > 1 million animals) until 1976.

The model suggests (line a in Figure 9) that the Serengeti ecosystem should remain stable in the future provided that the rainfall will be within the range of that experienced historically and provided that Mau forest remains forested as in the 1970s (Figure 2), and that it should remain stable in the face of climate change (Gereta et al., 2002).

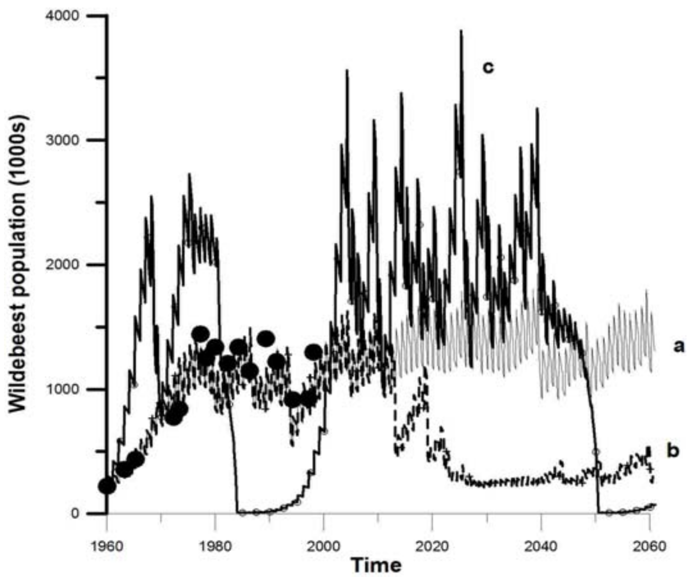


Figure 9. Time series plot of the (●) observed and predicted wildebeest population for historical rainfall patterns in the year 1960-2006 repeated in the future and for (a) 1972 hydrologic conditions in the Mara River, (b) 2006 hydrologic conditions in the Mara River, and (c) if the animal migration was stopped by providing drinking water throughout the ecosystem.

The model suggests that if the droughts of 1948 or 1972 would occur again, with the present levels of deforestation of the Mau forest and of use of Mara River water for irrigation in Kenya, the Mara River would dry out; as a result the herbivore population would collapse from the lack of drinking water (line b in Figure 9). This model also suggests that providing drinking water to the animals at artificial water holes spread throughout the ecosystem would lead to decadal time-scale booms and busts of the herbivore population (line c in Figure 9). The model thus suggests that the Serengeti ecosystem stability is maintained by the annual migration that partitions the ecosystem in rainfall-fed, seasonally used, compartments, and that it requires a perennial Mara River to survive droughts. It is thus necessary to maintain perennial flow in the Mara River, and this requires remediation measures in Kenya.

10. THE MARA RIVER BASIN HUMAN HEALTH STATUS

With a rapidly growing human population all along the Serengeti ecosystem, human and wildlife interests are increasingly interconnected. It is important to use socio-economic benefits from the Serengeti to improving human health. According to surveys that covered the three districts through which the Mara River traverses in Tanzania, the most common water related diseases and their average levels of incidence are malaria 40%, schistomiasis 17%, diarrhoea 16%, dysentery 8%, typhoid 5%, and skin infections 2% (Serengeti District Health Profile Report of 2001).

Malaria (water-related), schistomiasis (water-based) and diarrhoea (water-borne) are thus the most threatening diseases in this area. Mortality rates of malaria and diarrhoea for infants up to 5 years of age reach 19% and 5% respectively.

If the present mismanagement of the Mara River in Kenya continues unhindered and the river stops flowing, water will only remain in stagnant pools. This is likely to increase the incidence and mortality rates of some diseases, particularly diarrhoea, dysentery and skin infections because they are associated with personal and general hygiene. The formation/creation water pockets, instead of water flowing in the river, would provide a favourable environment for the spread of diseases like malaria and Schistomiasis.

The Tanzanian Government's water policy is intended to ensure proper protection and equitable use of water sources for both social and economic development for the benefit of all communities. The Government has been involving various communities in the processes of planning, selection of appropriate technology, construction, contribution and management of water projects. In addition, the Government has been encouraging the participation of institutions such as TANAPA, non-governmental organizations, private institutions, religious organizations and the public in improving the provision of water services in the country. Provision of safe and clean water has reached 50% of rural population and 73% of urban population. Rural water supply efforts that have been taken of the Government in this connection include construction, expansion and rehabilitation of existing water projects as well as exploration of new water projects in some regions. Efforts have also been made to promote rainwater-harvesting technology as an extra source of water in some areas. In addition, communities have been sensitized to manage and rehabilitate their water projects through training, seminars and workshops.

11. PAPYRUS RESTORATION DEMONSTRATION PROJECT

Rubondo Island National Park (RNP) in the south-western corner of Lake Victoria has totally protected papyrus wetlands. Outside the national park these sites have been destructed in large scale, and the remaining are in danger of disappearance if the current trends continue. Subsequently, many fishing camps are deliberately located very near the protected area and heavily populated to try to capture more fish as they move outside the park, and even illegally fish inside the park boundaries. RNP is unceasingly conducting anti-poaching patrols and poaching is unrelenting. To provide a long term solution, RNP is now promoting papyrus wetlands rehabilitation outside the park in collaboration with Nkome village in Geita district to stimulate the fisheries outside the park, reduce the fish poaching inside the RNP, raise the income of the local people, while using these constructed wetlands will help clean the waste waters entering the lake from the catchments. The socio-economic situation on the ground is difficult as rapid human population growth, environmental destruction, fisheries decline, food insecurity and poverty are pressing issues of the day. Papyrus wetlands are generally not appreciated ecosystems. Misuse of the papyrus wetlands range from farming all the way to the waterline, washing clothes, dishes and bathing in the lake, pasturing domestic animals, clearing and burning, establishment of settlements, waste dumping sites and poisoning. Beach Management Units (BMUs) working under the Fisheries Department generally has the mandate to oversee on the protection of the beaches and control of fisheries in their respective areas. In practice, except for RNP, Lake Victoria is entirely open access as far as the fisheries and the wetlands are concerned.

The rehabilitation project was implemented during a time of conflicting policies – people were encouraged to farm the wet areas during drought periods, conflicting with the Fisheries policy which protected the area for fish habitats. In Tanzania there is no wetlands policy. Any serious wetland conservation work would likely face opposition from the local communities through farmers who want the area for farming, fishermen who use illegal fishing methods, and the general population who wants wetlands products for different uses. However, evaluation of the situation around RNP shows that there is high potential for cooperation from the communities if conservation education is given and support to wetlands conservation/rehabilitation is granted at the local levels. This is what our pilot project aimed to achieve. Observation shows that there are local people who are well equipped with the skills of rehabilitation/restoring wetlands. Their experience and collaboration is important for scientific wetland rehabilitation projects. A populated village 10 km from RNP was selected as a pilot site because the readiness of the community to engage in environmental conservation and management, as well as a strong community leadership to oversee on it. The local people will ultimately take ownership in the prospect of earning an income in a few years time from fisheries and biofuels. The rehabilitation site is an old wetland that had been destroyed. It is sheltered from strong winds and sloping gently. It is located near the local market and near the ship landing site. Hundreds of people flood the vicinity of the area on Mondays and Thursdays, which are market days, and thus the site has an important educational value.

Villagers removed invasive vegetation from the plot, and then the earth, compacted by human activities, was dug by a tractor. Spoon-shaped creeks were constructed that run across the swamp from high to low elevation to bring water and small fish to most parts of the plot. We planted *Cyperus papyrus* spaced 1m x 1m apart, thus creating a papyrus fringed habitat for fish. We carefully included some substrate material when transplanting the papyrus so as to bring to the new site meiofauna and invertebrates to seed the new created swamp. Other species planted include *Typha sp*., *Mimosa sp*. Half of the land was not in water except in the built-in creeks; when the water level rises, the wetland can migrate there naturally. In these dry areas, we planted grass species, to attract worms and other small animals which are food for fish. The grasses were placed at the back of the plot so as to ensure maximum utilization of the whole plot by fish. We fenced the area, placed a signboard to warn people to keep livestock away. An MoU was drafted between RNP and Nkome village, which was to take ownership and responsibility of removing invading hyacinth, guarding the area, and managing it while earning the income after about a year when the papyrus matured. RNP provided scientific expertise and supervise the harvest of above ground papyrus biomass for biofuel for the villagers. A year into the project the papyrus vegetation was healthy and expanding, bird life had returned, juvenile tilapia fish were abundant, the vegetation was harvested as biofuel and the swamp allowed to regenerate. Several other villages have already requested a similar papyrus wetlands rehabilitation project in their area.

12. THE LINK BETWEEN INTERNAL WAVES, EUTROPHICATION, FISH KILLS, AND PAPYRUS

Lake Victoria is density stratified, with warm, oxygenated water at the surface and colder, anoxic waters at the bottom (Newell, 1960; Myanza, unp. data). The upper layer, the epilimnion, is separated by a transitional layer, the thermocline, from the lower layer, the hypolimnion. The thermocline of Lake Victoria was at 40 m depth in the 1960s (Newell, 1960) and 20 m at present (Okaronon and Wadanya, 1991; Myanza *et al*., 2006; Rutagemwa *et al.*, 2006).

Strong wind induces internal waves in the thermocline (interface), disturbing its position (Wetzel, 1983; Myanza *et al.*, 2006). In the main part of Lake Victoria the lake is open water with no islands to block the flow of wind fetch. There the wind causes a surface set-up downwind and a surface set-down upwind, causing the interface to fall downwind and rise upwind. An upwelling occurs at the upwind side of the lake. Past researchers have noted the internal waves of Lake Victoria from occasional cross-lake transects, but the measurements at monthly or quarterly intervals are too infrequent to quantify the dynamics and importance of these waves. It is known however that these waves can bring nutrients and anoxic waters to the surface (COWI, 2002, Gikuma-Njuru and Hecky, 2005). The anoxic waters cause fish kills (Wetzel, 1983; Ochumba and Kibaara, 1989; World Bank, 1996).

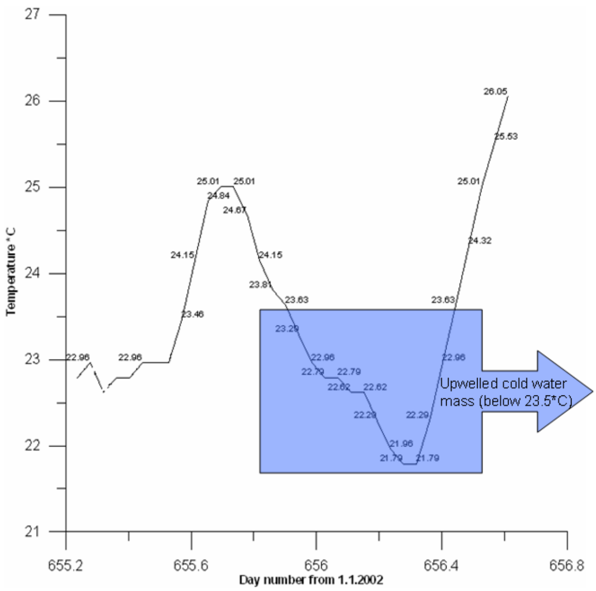
These waves may be wind-driven. Assuming that the lake is a two-layer system with a top and bottom layer separated by a sharp thermocline, with a top layer of thickness h_1, the wind over the lake will downwell the thermocline on the downwind side of the lake and lift the thermocline towards the surface on the upwind side. The wind required for the thermocline to surface can be estimated from the Wedderburn number (Stevens and Lawrence, 1997)

$$W = g\, h_1^2\, \Delta\rho \,/\, L\, u_*^2\, \rho \tag{1}$$

where g is the acceleration due to gravity, L is the length of the lake along the wind direction, u_* is the wind-driven kinematic shear stress in water, ρ is the density of water and $\Delta\rho$ is the change of density across the thermocline. Upwelling occurs when $W < 1$ (Stevens and Lawrence, 1997; Kaplan *et al*., 2003; Talavera and Richardson, unp. data).

For Lake Victoria, the thermocline is predicted to surface under a sustained wind of constant direction for several days with a speed of at least 5 m s^{-1}. Since a westward wind of such speed is common, an upwelling should occur commonly on the east side of the lake. This sustained upwelling is not observed. The reason is the land-lake breeze which prevents a recirculating flow to develop near the shore in open waters (COWI, 2002).

However, not all of the lake is open waters. Around RNP, the islands form an archipelago that blocks the large-scale circulation and the Wedderburn number approach is not applicable. A situation similar to that studied in oceanic islands (e.g. Wolanski *et al.,* 2004) may apply. Island-generated internal waves do not travel around the island and upwelling is topographically-driven and occurs at local scales especially around headlands. The difference between these two upwelling mechanisms is huge: the wind-driven lake-wide internal wave is basin-scale, i.e. tens of km. The topographic internal wave is at the scale of an island, i.e. a few km. Such internal waves were measured at RNP from 2002 to 2006 by using temperature data loggers (Figure 10; see also Kiwango, 2007). Upwelling typically occurs for about 200 hours in a year at one location; the thermocline reached near the surface for several days during a major upwelling event; the upwelling events were localized and patchy. This island-generated patchiness makes it difficult to characterize the upwelling dynamics in the southwest region of Lake Victoria where islands are scattered widely. The historical minimum temperature value for Lake Victoria as measured at 60 m depth and reported by Lehman et al (1998) is 23.5 °C observed in 1990 and 1992. Such low temperature, characteristic of deep Lake Victoria waters, were observed at five of the six measurements sites at Rubondo Island for the total of 679 hours (i.e 1.3 % of the time) in 2002-2006. In fact, we observed the coldest water ever reported for Lake Victoria during an upwelling that lasted for 12 hours (Figure 10).



Day number from 1.1.2001	Temperature in °C
655.81944433329	24.15
655.86111099996	23.81
655.90277766663	23.63
655.94444433329	23.29
655.98611099996	22.96
656.02777766663	22.79
656.06944433329	22.79
656.11111099996	22.62
656.15277766663	22.62
656.19444433329	22.29
656.23611099996	21.96
656.27777766663	21.79
656.31944433329	21.79
656.36111099996	22.29
656.40277766663	22.96
656.44444433329	23.63
656.48611099996	24.32
656.52777766663	25.01
656.56944433329	25.53
656.61111099996	26.05

Figure 10. Time-series plot of the temperature at 2 m depth Kasenye1, RN. Coldest water mass for this study was observed at 21.79 °C for 2 hours. The upwelling lasted for 12 hours.

Even when the internal wave does not reach the surface, it brings deep, nutrient-rich, anoxic waters close to the surface within the reach of wind-mixing. Thus some nutrients become available for productivity in the biological active surface layer of the lake water. Such a process is beneficial in oligotrophic lakes where the surface layer is depleted of nutrients, such as Lake Villarria in central Chile (Meruane *et al*., 2007) and Lake Tanganyika (Naithani *et al*., 2007). But for Lake Victoria, which is already threatened by severe eutrophication problem, this upwelling exacerbates the problem because it recycles back to the surface layer the nutrients that were lost from that surface layer and stored in the bottom layer.

When anoxic, bottom waters are upwelled by internal waves into surface waters, they shock and possibly kill large fish that do not quickly react and swim to the surface to seek oxygen-rich water. Such fish kills were observed in patches around Rubondo and in the south-western part of Lake Victoria almost every year since the early 1980s (Kaoneka and Mlengeya, unp. data; TANAPA, unp. data; Rutagemwa *et al*., 2006; Borner, pers. comm.). Fish die-offs are mostly of Nile Perch, which surfaced quickly from deeper water in large numbers and died. Usually the stomach (not the swim bladder) was inverted and pressed out of their mouths. Die-offs in shallow waters were also witnessed, which affected other organisms as well, including tilapia. Poison as a cause of the fish kills was ruled out by laboratory investigation of the dead fish samples taken to Veterinary Investigation Center in Mwanza (Kaoneka and Mlengeya, 2000).

13. THE IMPORTANCE OF ANNEXING SPEKE GULF INTO SERENGETI NATIONAL PARK

The 5 km wide Speke Gulf Game Controlled Area (SPGCA) separates Lake Victoria from the western corridor of the SNP (Figures 1a and 1b). The SPGCA is bordered by Robana and Mbalageti Rivers to the north and south respectively. The shore of the Speke Gulf is typically a wetland which supports dense vegetation including papyrus and typha communities. Currently there are two villages located within the Speke Gulf GCA and the people are engaged in a range of economic activities including agriculture, fisheries, tourism, livestock keeping and small–scale trade. Wildlife hunting is not permitted. Given that the Mara River may dry out in the future due to water resources mismanagement in Kenya, Speke Gulf provides an alternative permanent source of water for wildlife.

Wild animals (especially elephants) have for years continued trespassing on these village in SGGCA in search of water Available evidence indicates increased human – wildlife conflicts of various nature and dimensions, including human death. As human population and activities continue to expand in the area, so are the chances of conflicts from wildlife in search of water from Lake Victoria and therefore it is only logical that separate one population (wildlife or humans) from another. TANAPA is considering resettling the villages outside the SGGCA and annexing the area in the SNP (Figure 1b). While this may not solve the problem of a lack of water in the northern region of the SNP should the Mara River dry out in the future, it will lessen the negative impact of such a man-made drought on the Serengeti ecosystem.

There are additional socio-economic benefits for this annexation. It would ease human - wildlife conflicts and protect papyrus wetlands breeding grounds for fish (Mnaya *et al*., 2006;

Kiwango and Wolanski, 2008), replenishing fish stocks in the nearby village fishing grounds and guaranteeing food security for the empoverished local population. Also, this annexation will also facilitate lake–based tourism, offering both rich terrestrial and aquatic tourism experiences. This would diversify tourism products for the SNP in the line with the Serengeti National Park General Management Plan (2006).

14. The Management of International Waters

The future of the Serengeti ecosystem depends on solving water resources management issues of both Lake Victoria and the Mara River. For Lake Victoria the key issues are the overdrawing of water by Uganda as well as the conservation of the papyrus wetlands by Tanzania, Uganda and Kenya, including sustainable harvesting to combat eutrophication. For the Mara River the key issues are deforestation of the Mau forest in Kenya and overdrawing of Mara River water for irrigation in Kenya. Thus the future of the Serengeti ecosystem is very much an international issue. To guarantee the future of the Serengeti ecosystem, a transboundary management plan is needed for both Lake Victoria and for the Mara River, compatible with ecohydrology principles. There is no legal mechanism for this.

Currently, there are efforts encouraged by Norway to establish an authority that will manage and ensure sustainable utilization of Nile basin waters. In February 2002 a meeting of Ministers responsible for water sector was held in Cairo to discuss a draft on institutional and legal system to oversee the utilization and development of the Nile Basin waters. The Tanzanian Government through the Ministry of Water and Livestock continues to follow up on efforts to establish the Nile Basin Authority. Furthermore, the Ministry is proposing to use this Nile River template to establish a legal institution for the development of the Mara River Basin. While this issue has been discussed by the East African Community, there has been no support from Kenya to establish such a legal Authority. There is little room for optimism so far because, while the deforestation of the Mau forest has slowed down, irrigation is increasing and Mara River water is over-allocated in Kenya (Nile Basin Initiative, 2008). Indeed it can be feared that no international agreement is likely for the Mara River in view of the example of Uganda unilaterally deciding to overdraw water from Lake Victoria since 2000 over what was agreed by an international agreement (‘The Agreed Curve’) to increase hydroelectricity production and as a result to decrease the water level by more than 2 m (see lines a and b in Figure 3; Kiwango and Wolanski, 2008), without considering the environmental and socio-economic consequences for Kenya and Tanzania who also share the lake. There is thus no precedence in the East African Community for a country to have to consider its neighbours in using international waters. In addition, the experience in East Africa (e.g. Mtahiko et al., 2006) shows that, as long as the true cost of water is not charged to users – commonly cost recovery rates in ***irrigation*** amount to only 20-25 % of true costs (Lutz et al., 1999) - irrigators will have no qualm about drying out a river. There is no clear mechanism at present to prevent this scenario to develop in the Mara River, given the ambiguities in Kenya’s Water Act 2002 about the need to maintain environmental flows for the whole river, and given the limitations of technical and financial resources facing Kenya’s Ministry of Water (Mumma, 2005).

The fate of the Serengeti ecosystem remains on a knife's edge until a water management plan for Lake Victoria and the Mara River, based on ecohydrology principles, is signed by Uganda, Kenya and Tanzania.

The fate of Lake Victoria ecosystem is even more tenuous because it involves land-use over much of Tanzania, Kenya, and Uganda. As the lake eutrophication increases due to poor land-use in the catchment area, the anoxic layer will continue rising towards the surface, from > 50 m in 1961 to as low as 35 m in 1990, and 20-30 m in 2005 (Hecky *et al.,* 1994, 1996; Kaufman, 1992; Lehman *et al*., 1998; Mugidde, 1993; Ochumba and Kibaara, 1989; Reinthal and Kling, 1994; World Bank, 1996; Rutagemwa *et al.,*.2006). The anoxic layer being closer to the surface, the internal waves will more frequently upwell nutrients to the surface waters, thus accelerating the eutrophication of Lake Victoria.

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Chapter 7

TEMPORARY STREAMS: THE HYDROLOGY, GEOGRAPHY, AND ECOLOGY OF NON-PERENNIALLY FLOWING WATERS

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ABSTRACT

Temporary streams represent a significant yet understudied and particularly vulnerable portion of river networks. While the vast majority of stream and river research to date has focused on perennial flowing waters, recent work reveals that temporary streams are not only abundant and widely distributed, but also play a significant role in the hydrological and ecological integrity of lotic networks. In this chapter, we seek to summarize the current state of the science of these ubiquitous portions of river networks while simultaneously stressing the need for their future investigation. We begin by defining temporary streams and their hydrology and highlighting their abundance and extent. We then consider the ecological significance of temporary streams, including their role as faunal and floral habitat providers, biogeochemical processors, and connectivity corridors within river networks. The chapter concludes with a discussion of policy issues surrounding temporary streams and the anthropogenic disturbances they face.

INTRODUCTION

Temporary streams are channels that lack surface flow during some portion of the year. Positioned at the interface between fully aquatic and fully terrestrial ecosystems, they are among the most abundant, widely distributed, and dynamic freshwater ecosystems on Earth

[Comin and Williams 1994; Poff 1996; Williams 1996; Larned et al. 2010]. Cumulatively, they account for a significant proportion of the total number, length, and discharge volume of the world's rivers [Dodds 1997; Tooth 2000; Larned et al. 2010]. Temporary streams range in size from the smallest of episodically flowing zero-order rivulets with small drainage basins, to seasonally flowing headwaters, to higher order river reaches that are spatially disconnected at some portion of the year.

Research to date suggests that these temporary waters are not only abundant, but ecologically valuable [Meyer et al. 2003; Nadeau and Rains 2007; Larned et al. 2010], serving as animal and plant habitat, zones of nutrient and carbon processing, and connectivity corridors intimately linked to both the watersheds they drain and the river networks to which they are episodically connected. Despite periodic discontinuities in surface flow, temporary streams are intimately linked - both hydrologically and ecologically - to their watersheds and perennial waters [Cummins and Wilzbach 2005; Nadeau and Rains 2007]. Yet, temporary streams have been "historically neglected" by scientists and society at large [Larned et al. 2010]. They are understudied relative to continuously flowing perennial streams [Robson et al. 2008], poorly mapped [Meyer and Wallace 2001], and faced with numerous anthropogenic disturbances [Dodds et al. 2004; Brooks 2009]. Moreover, it is expected that streams will become increasingly more temporary due to global climate change [Lake et al. 2000; Palmer et al. 2008; Brooks 2009]. Increased awareness and knowledge of their extent and integral ecologic and hydrologic role in river networks should aid in their management and protection. Here, we review the current state of knowledge about temporary stream hydrology, geography, and ecology and demonstrate their essential role within river networks. We review the anthropogenic disturbances they face and the challenges with respect to protecting them.

HYDROLOGY

Hydrology is perhaps the most fundamental driver of physical, chemical, and biological processes in streams and is often considered a "master variable" controlling geomorphology, substrate stability, faunal and floral habitat suitability, thermal regulation, metabolism, biogeochemical cycling, and the downstream flux of energy, matter, and biota [Power et al. 1988; Resh et al. 1988; Poff and Ward 1989; Poff 1996; Poff et al. 1997; Dodds et al. 2004]. Flow magnitude, frequency, duration, timing, and rate of change together characterize a stream's *flow regime* [Poff et al. 1997]. Unlike larger streams and rivers, temporary streams have a flow regime defined by periodic drying and wetting that places them at a unique interface between fully terrestrial and fully aquatic environments. Their position at this terrestrial-aquatic ecotone sets them apart from continuously flowing portions of river networks in that they can support both land- and water- based ecosystem functions and services.

Streams may be defined according to their surface hydrologic flow duration as either perennial or temporary (also known as "non-perennial") [Hansen 2001]. Under normal circumstances, *perennial streams* flow throughout the year, whereas *temporary streams* lack surface flow for some portion of the year. Temporary streams are classified as either intermittent or ephemeral. *Intermittent streams* flow seasonally in response to snowmelt and/or elevated groundwater tables resulting from increased periods of precipitation and/or decreased evapotranspiration. The groundwater table is above the bed of an intermittent stream during some portions of the year (Figure 1c) and below it during others (Figure 1d). During periods of seasonal surface flow, the groundwater table is above the bed, and the intermittent stream receives a baseflow supply whereby it is a *gaining stream* – that is, it gains water from groundwater (Figure 1c). During dry seasons and/or dry conditions, the groundwater table is below the channel bed, and the channel lacks a source for baseflow (Figure 1d). Under these low groundwater table conditions, the intermittent channel is a *losing stream* – that is, it loses water to groundwater (Figure 1d). *Ephemeral streams* are those that only flow during and in immediate response to precipitation events. The groundwater table is situated below the streambed of an ephemeral stream throughout the entire year such that the channel never receives groundwater discharge and, in turn, lacks a baseflow source (Figure 1e,f). As a result, ephemeral streams are always losing streams.

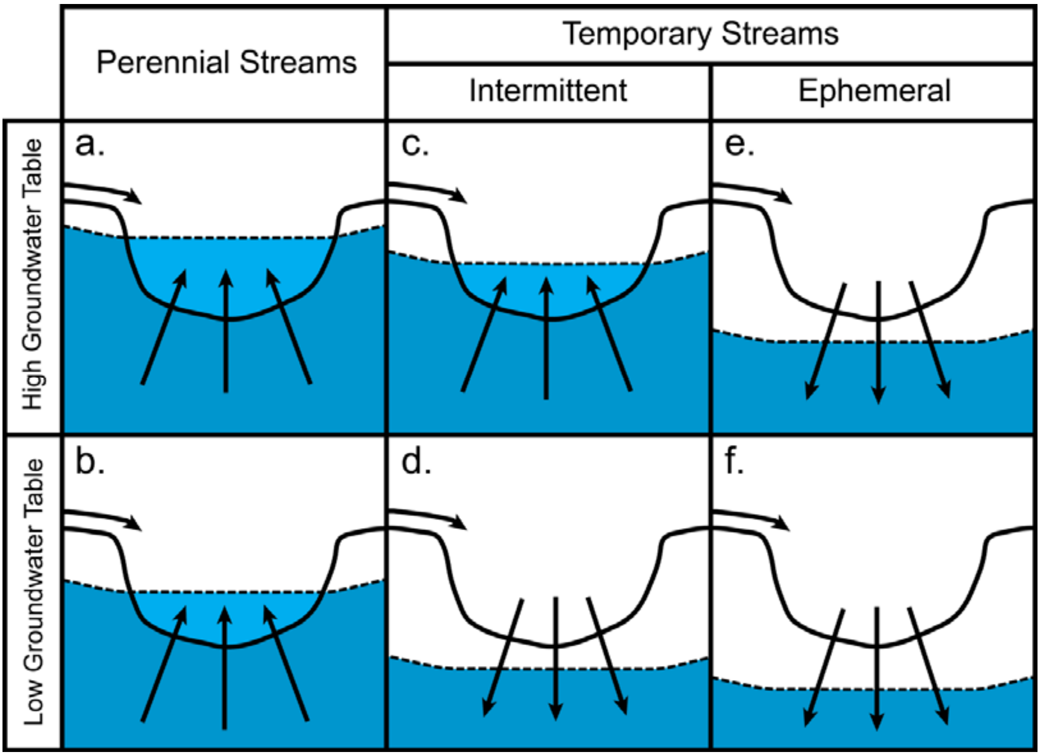

Figure 1. Channel cross-sectional schematic showing perennial, intermittent, and ephemeral streams under high and low groundwater table conditions. Dashed line indicates groundwater table elevation. Arrows indicate surface water and groundwater flowpaths. a) Perennial – High Groundwater: gaining stream. b) Perennial – Low Groundwater: gaining stream. c) Intermittent – High Groundwater: gaining stream. d) Intermittent – Low Groundwater: losing stream. e) Ephemeral – High Groundwater: losing stream. f) Ephemeral – Low Groundwater: losing stream.

Temporary streams may also be spatially discontinuous where surface water is present at some reaches of an individual tributary, but absent at others. The result is a pattern of longitudinal surface water patchiness along the tributary. Such fragmentation may result from drying and subsequent hydrologic isolation of pools along a tributary or downwelling portions of the streambed where all surface flow is subducted to subsurface flow, or *hyporheic*, pathways before re-emerging in upwelling zones at some distance downstream. Drying in a temporary stream with a pool-riffle geomorphology begins with reduced flow through the reach. As drying persists, water levels become lower, surface flow ceases in riffles altogether, and a series of separated pools results. Depending on the severity of drying as well as depth of pools, surface water may be lost entirely [Labbe and Fausch 2000] (Figure 2). Longitudinally isolated pools are especially common in arid climates and regions defined by strong seasonal wet and dry periods. Anthropogenic groundwater abstraction for municipal, agricultural, and industrial use may also lead to lowered regional groundwater tables and subsequent longitudinal isolation of pools along rivers [Dodds et al. 2004]. Even in the absence of surface flow, temporary streams may contain hyporheic flowpaths that serve as habitat refugia from drying, zones of biogeochemical processing, and hydrologic connections to downstream waters.

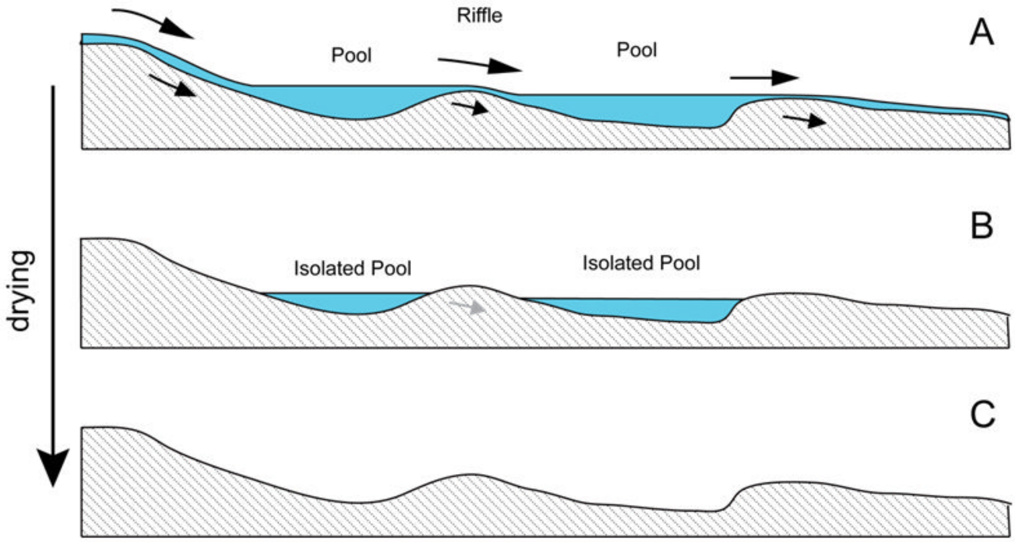

Figure 2. Contraction of a stream reach under increasingly dry conditions. Arrows indicated surface and groundwater flowpaths. A) Surface hydrologic connectivity exists throughout the reach such that pools are connected via riffles. B) As drying persists, riffles dry and pools contract until they are geographically isolated. C) If drying persists long enough, all surface water may be lost to either groundwater reserves or evapotranspiration.

GEOGRAPHY

Abundance and Extent

Temporary streams are among the most abundant, widely distributed, and dynamic freshwater ecosystems on Earth [Comin and Williams 1994; Poff 1996; Williams 1996; Larned et al. 2010]. Collectively, they account for a significant proportion of the total number, length, and discharge volume of the world's rivers [Dodds 1997; Tooth 2000; Larned et al. 2010]. While they are most abundant in arid and semi-arid regions, temporary streams are also commonly found throughout the globe between 84 deg N and 84 deg S latitude [Larned et al. 2010].

Ephemeral and intermittent streams are often positioned at the headwaters of river networks [Dodds et al. 2004] (Figure 3). Headwaters are formed by watersheds draining small parcels of land and, in turn, small volumes of water. As a result, headwaters are more susceptible to drying than downstream reaches because they have smaller drainage areas with less recharge potential for baseflow maintenance [McMahon and Finlayson 2003; Fritz et al. 2008]. Moreover, headwater catchments are often less pervious than downstream portions of watersheds, in turn resulting in minimal storage capacity to maintain baseflow [Burt 1992] and subsequent formation of temporary streams [Dodds et al. 2004; Levick et al. 2008; Brooks 2009] ranging from small ephemeral rills and gullies to larger, more well-developed intermittent channels. A study in Chattahoochee, Tennessee (USA) found, for instance, that 78% of the stream reaches within a watershed were headwaters, and that a majority of those headwaters were temporary [Hansen 2001]. In temperate regions, summer drought conditions can result in "summer-dry" headwaters. With increased precipitation and/or decreased evapotranspiration, dry headwaters rewet and account for a substantial portion of the drainage network. In fact, the temporary portion of river networks may often exceed the length of permanently flowing reaches [Dieterich and Anderson 1998].

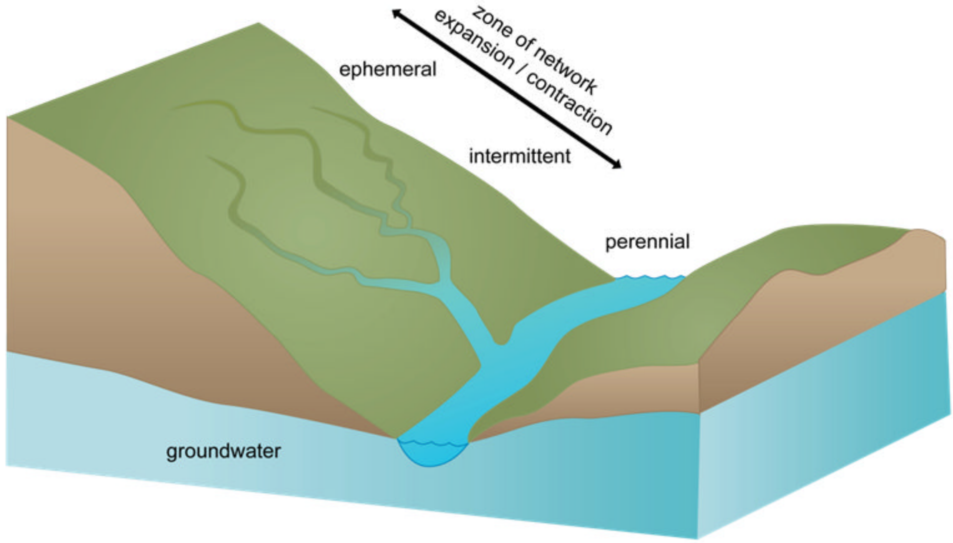

Figure 3. Typical transition from temporary to perennial streams at the headwaters of a river network. Ephemeral and intermittent reaches are a zone of network expansion under wetting conditions and contraction under drying conditions. [Modified from symbols courtesy of the Integration and Application Network (ian.umces.edu/symbols/), University of Maryland Center for Environmental Science].

Headwater rewetting results from a combination of increased soil saturation, shallow subsurface flow, a general rise of the groundwater table, and subsequent expansion of the active drainage network. Gomi et al. [2002] refer to the headwaters of river networks as "transitional channels." The transitional channels are zones of drainage network expansion and contraction under wet and dry conditions respectively (Figure 3). In mesic regions with a more continuously wet climate, transitional channels may be short, ephemeral rills that flow in immediate response to heavy rain events. The length of individual temporary channels is likely to be much longer in more xeric climates, including desert and prairie stream networks [Dodds et al. 2004].

Although a global inventory of temporary streams has not yet been compiled, several national and regional estimates exist and collectively underscore their abundance. Headwater streams cumulatively account for the greatest portion – perhaps as much as 80% - of stream length within river networks [Leopold et al. 1964; Meyer et al. 2003; Lowe and Likens 2005]. In the United States (excluding Alaska), it is estimated that ephemeral and intermittent streams, many of which are located at the headwaters, total 3,200,000 km of stream length, or nearly 60% of cumulative stream length [Nadeau and Rains 2007]. In the arid and semi-arid southwestern United States, temporary streams account for > 80% of the entire network [Levick et al. 2008]. Over 50% of the Australian mainland is drained by temporary streams [Williams 1983] (Figure 4), much of which are located in the southeast of the continent [Lake et al. 1986]. Due to the marked wet/dry seasonal precipitation pattern in the Mediterranean climate, nearly half of the river network in Greece is temporary [Tzoraki and Nikolaidis 2007]. With respect to specific watersheds, Hansen [2001] found that > 70% of the Chattooga River network (southeastern USA) is temporary, and Doering et al. [2007] estimated nearly 50% of the Tagliamento River drainage (northeastern Italy) does not flow continuously.

Figure 4. Brachina Creek, Flinders Ranges (South Australia) – one of countless temporary streams draining the Australian mainland. [Photo: A. Boulton].

In addition to headwater channels of river networks, some higher order branches – perhaps 4th and even 5th order [McBride and Strahan 1984] – exhibit temporary surface flow. Not surprisingly, temporary streams are common in more arid climates (e.g., desert and prairie streams) or regions with a pronounced dry season (e.g., streams in Mediterranean climates) [Gomez et al. 2009]. As a region enters a period of drought or a dry season, surface water is lost to both evapotranspiration and groundwater storage. The result is a retreating stream "front" characterized by longitudinally discontinuous surface flow [Larned et al. 2010]. Riffles dry and pools contract such that they become geographically separated [Fisher et al. 1982; Stanley et al. 1997] (Figure 2). Upon return of wetter conditions – a result of increased precipitation, decreased evapotranspiration, decreased groundwater abstraction, or some combination therein – an advancing stream "front" reconnects the previously fragmented tributary. In arid regions, a single large rain event can rapidly reconnect the network [Gomez et al. 2009].

Mapping

It has long been recognized that commonly used stream maps grossly underestimate actual stream length [Mueller 1979; Meyer and Wallace 2001]. The scale at which drainage networks are mapped can dramatically affect the total number and length of streams that are identified. With decreasing map resolution, fewer and only increasingly larger streams tend to be indicated [Miller et al. 1999]. Temporary streams are often individually small and therefore the most likely to be omitted as map resolution decreases [Roy et al. 2009]. Meyer and Wallace [2001], for instance, showed that the total stream length within the Coweeta Creek watershed (North Carolina, USA) on a coarse resolution 1:500,000 scale topographic map was only 3% of the length indicated on a standard United States Geological Survey (USGS), higher resolution 1:24,000 scale map. They also found that lower order streams (i.e., headwater, intermittent, and ephemeral streams) were more likely than higher order perennial channels to be unmapped at the coarser resolution.

Yet even 1:24,000 scale maps – the most common source of drainage network data in the United States - may still omit the uppermost reaches and thereby significantly underestimate total stream length within a watershed. Studies in temperate North America have shown that 1:24,000 scale maps can exclude nearly all temporary streams, resulting in >70% omission of actual stream length [Hansen 2001; Heine et al. 2004; Roy et al. 2009]. Studying the Arroyo de los Frijoles catchment in the arid southwestern United States, Leopold et al. [1964] found that contour patterns resulted in a network with nearly 260 first order channels, yet 1:24,000 scale maps identified no stream channels within the entire watershed. The researchers further identified 86 ephemeral channels in just *one* of the first order streams by physically walking its length [Leopold et al. 1964].

ECOLOGICAL SIGNIFICANCE

Because temporary streams exist at an interface between aquatic and terrestrial ecosystems they represent an "intimate ecological linkage" between the stream and its watershed [Cummins and Wilzbach 2005]. They provide valuable habitat to a wide variety of plant and animal species and function as biogeochemical hot spots that retain, process, and transform carbon, nutrients, and particulates. They are hydrologically and ecologically linked to perennial and other temporary waters with which they exchange matter, energy, and organisms.

Habitat Provision

It has long been recognized that temporary streams support a wide diversity of life [Stehr and Branson 1938]. Positioned at a terrestrial-aquatic ecotone, they provide unique habitat [Meyer et al. 2007]. Fauna and flora, both terrestrial and aquatic, have developed adaptations and life histories to cope with these hydrologically dynamic reaches where habitat can shift rapidly from high velocity, well-oxygenated riffles to stagnant, isolated pools to completely dry streambed and back. Among the many factors controlling biological communities in temporary streams, patterns of flow may be the most important [Poole et al. 2006]. Both animals and plants are faced with periodic and often rapid disappearance and reappearance of vast areas of habitat in streams that alternate between wet and dry states.

ANIMAL HABITAT

Temporary streams support a variety of fauna including macroinvertebrates, fishes, amphibians, and streamside mammals, reptiles, and birds. Whether serving as sites of oviposition, spawning, rearing, refugia from drying, or dietary hot spots, temporary streams harbor a wide diversity of animal life. While these channels undoubtedly harbor a rich diversity of micro- and meiofauna (e.g., microbes, biofilms, rotifers, crustaceans), here we focus our discussion on macrofauna habitat provision and adaptations in temporary streams.

Macroinvertebrates

While macroinvertebrate abundance and taxa richness are generally lower in ephemeral and intermittent streams relative to permanent waters [Clifford 1966; Williams and Hynes 1976b; Williams 1996; but see Flinders and Magoulick 2003], temporary streams may also support rare or unique species [Dieterich and Anderson 2000]. Lower species richness is widely attributed to the more extreme conditions and variable habitat found in temporary streams [Boulton and Suter 1986] and may also be related to reduced aquatic habitat area as the result of drying (i.e., species-area effects) [Lake 2000]. Studying stream intermittency and macroinvertebrate assemblage diversity in the Great Plains (central USA), Fritz and Dodds [2005] developed a stream harshness index based upon a suite of hydrologic flow regime variables. They found that harshness indices were high in intermittent streams, low in perennial streams, and negatively related to macroinvertebrate diversity and species richness [Fritz and Dodds 2005].

Although they lack continuous surface water, temporary streams may harbor robust and/or endemic aquatic or semi-aquatic macroinvertebrate communities [Feminella 1996; Dieterich and Anderson 2000; Stout and Wallace 2003; Robson et al. 2005; Williams 2005; Collins et al. 2007] and often share a common pool of macroinvertebrate taxa with perennial streams [Boulton and Lake 1992, Feminella 1996, del Rosario and Resh 2000, Shivago 2001, Smith et al. 2003, Collins et al. 2007, Arscott et al. 2010]. Moreover, some studies have found that temporary streams harbor a unique set of macroinvertebrate taxa not found in nearby perennial reaches [Feminella 1996; Dieterich and Anderson 2000]. Other studies, however, have found little or no difference between temporary and perennial stream macroinvertebrate communities [Beugly and Pyron 2010; Robson et al. 2008].

Why do some temporary streams boast uniquely adapted macroinvertebrate assemblages while others do not? One study attributed the lack of temporary stream-adapted macroinvertebrates to a lack of nearby permanent surface water refugia, length of drying period, and a lack of climate predictability [Arscott et al. 2010] whereby an unpredictable pattern of intermittency renders adaptation to drying unlikely. Several studies have reported general similarities between perennial and temporary streams with a limited number of taxa endemic to temporary reaches [Shivago 2001; Williams et al. 2004; Beche et al. 2006; Storey and Quinn 2008; Arscott et al. 2010]. It seems likely that rather than a duality where temporary stream communities are either unique to or largely shared with perennial streams, there exists a gradient of similarity depending on a variety of local conditions including hydrology, geomorphology, climate, competition, and predation.

Many macroinvertebrates - especially insects - in temporary stream reaches possess physiological and/or behavioral traits allowing them to persist even after surface flow has ceased [Williams and Hynes 1977; Towns 1985; Williams et al. 2004; Bonada et al. 2006; Storey and Quinn 2008]. Survival strategies used by temporary stream dwelling macroinvertebrate taxa are numerous, and Williams [1998] provides an excellent review of macroinvertebrate persistence strategies in temporary streams. Here, we discuss two strategies that are particularly important: 1) survival in refugia and subsequent colonization and 2) resistance to desiccation.

Survival in Refugia and Colonization

There are numerous permanent water refugia that macroinvertebrates use to persist when surface water disappears from stream channels. For example, some insects as well as miofaunal invertebrates have the ability to burrow down to the hyporheic zone [Stanford and Ward 1988; Palmer et al. 1992; Collins et al. 2007], though this may not be a viable survival tactic if drying occurs too rapidly and/or bed sediments are too coarse for burrowing [Boulton and Stanley 1995]. Additionally, isolated pools represent particularly valuable refugia from stream drying [Boulton 1989]. Special adaptations to survival in isolated pools may be required because these habitats often become depleted in dissolved oxygen over time. Researchers studying isolated pools along arid stream reaches have found chironomid larvae (non-biting midges) that utilize hemoglobin to tolerate low oxygen conditions [Williams and Hynes 1976; Boulton 1989; Stanley et al. 1994]. Some Ephemeroptera (mayflies) taxa are equipped with specialized gills to tolerate low oxygen levels [Boulton 1989; Miller and Golladay 1996]. Other macroinvertebrate taxa found in oxygen-depleted temporary streams can directly breathe atmospheric air. These include Coleoptera (beetles) [Boulton 1989; Stanley et al. 1994; Williams 1998], Hemiptera (true bugs) [Stanley et al. 1994; Williams 1998], and Tipulidae (craneflies) [Williams 1998].

Shorter cohort production intervals and rapid larval development enabling adults to disperse before severe channel drying allow macroinvertebrates to use nearby perennial waters as spatial refugia or terrestrial life stages as temporal refugia. The Plecopteran (stoneflies) *Riekoperla naso* accomplishes this by rapidly developing from an egg to a mature flying adult [Towns 1985]. Studying macroinvertebrate assemblages in a seasonally intermittent stream in the Great Plains (central USA), Fritz and Dodds [2004] identified the Ephemeropteran (mayflies) *Fallceon quilleri*, having an 18-day life cycle, and the chironomid (non-biting midges) *Cricotopus* sp., having a 6-day life cycle. Temporary stream dwelling Hemiptera (true bugs) larvae may mature quickly and abandon temporary reaches as flying adults [Williams and Hynes 1976]. Macroinvertebrates using such refugia can re-colonize temporary reaches quickly after surface flow resumes. Intermittent prairie stream-dwelling macroinvertebrate communities have been observed to reappear within one week of resumed surface flow [Fritz 1997].

Boulton [1989] sampled eight over-summering refuges including dried leaf litter, crayfish burrows, receding pools, a nearby permanent lake, rotting wood and bark, dry substrate from riffles and pools, and the hyporheic zone. Of the refugia surveyed, a majority of taxa were found in isolated pools, yet both pool and riffle substrata also harbored significant proportions of the total taxa. Overall, nearly 50% of taxa recorded were found in areas lacking freestanding water [Boulton 1989].

Resistance to Desiccation

Another adaptation to stream drying is the use of desiccation resistant life stages [Williams and Hynes 1976b; Williams 1996]. Evidence for survival as dormant eggs has been found for Diptera (flies) [Williams and Hynes 1976; Stubbington et al. 2009], Ephemeroptera (mayflies) [Boulton 1989], and Plecoptera (stoneflies) taxa [Dieterich and Anderson 1995]. Towns [1985] identified leptocerid caddisflies, which are known for depositing terrestrial egg masses, in an Australian temporary stream. Smith et al. [2003] found certain Trichoptera (caddisflies) taxa were able to aestivate as adults during dry periods. Boulton [1989] found that dormant water penny beetles under dry rocks became active after being submerged in water. Plecoptera (stoneflies), Trichoptera (caddisflies), and Ephemeroptera (mayflies) taxa may persist as embryos in dry streambeds, and some oligochaetes survive dry conditions as cysts [Williams and Hynes 1976]. Some chironomid (non-biting midges) taxa can survive drought in desiccation-resistant cocoons [Griswold et al. 2008].

Drought intolerant macroinvertebrates unable to locate viable refugia during dry conditions, however, may die and provide carbon and nutrient subsidies to riparian consumers, thereby linking temporary stream and terrestrial ecosystems [Williams 1987].

Fishes

Streams that lack surface water for some portion of the year may not seem like ideal habitat for fish species. However, when temporary reaches of a river network rewet, flowpaths re-emerge and allow for fish passage into once fragmented or disconnected segments. Under these conditions, fishes are able to migrate from perennial waters into newly accessible habitats, including floodplains and temporary reaches, where dietary resources may be largely untapped, competition can be low, and conditions amenable for spawning and juvenile rearing may exist [Erman and Leidy 1975; Erman and Hawthorne 1976; Hartman and Brown 1987; Junk et al. 1989; Dodds et al. 2004; Wigington et al. 2006; Colvin et al. 2009]. Murdock et al. [2010] observed large schools of fish inhabiting recently re-connected pool habitats in an intermittent portion of a prairie stream (Kings Creek, Kansas, USA) only 3 days after being reconnected via surface flow to a perennial reach. In Sycamore Creek (Arizona, USA), Stanley [1993] found that longfin dace (*Agosia chrysogaster*) built nests earlier in the breeding season and in greater numbers in recently rewetted intermittent stream reaches relative to perennial reaches, thereby suggesting a preference for the temporary reaches. Wigington et al. [2006] found that coho salmon (*Oncorhynchus kisutch*) smolts overwintered in intermittent reaches and were larger than their counterparts that overwintered in perennial reaches or river mainstems. Everest [1973] and Kralik and Sowerwine [1977] also found that intermittent streams often serve as critical refugia for juvenile salmonids during periods of high winter discharge in Pacific Northwestern (USA) streams.

In the upper Williamette River valley (Oregon, USA), agriculture and river regulation have reduced river floodplains to intermittent watercourses. Although these temporary waters are largely the result of anthropogenic modification of the watershed and stream network, 13 fish species – 90% of which were native – were found to inhabit them [Colvin et al. 2009]. Moreover, newly hatched and juvenile fishes were found in the intermittent streams, suggesting these seasonally available reaches offered conditions suitable for spawning and rearing [Colvin et al. 2009]. Yet, Colvin et al. [2009] also found that intermittent stream fish species richness decreased as distance of the intermittent stream to the closest perennial reach increased, suggesting that sites more distant from perennial waters may be less suitable fish habitat. Fish may take anywhere from hours to weeks to colonize newly re-wetted temporary stream reaches, largely depending on the distance of the temporary water to a permanent source [Larimore et al. 1959; Dodds et al. 2004]. Temporary stream reaches closer to perennial waters are likely to be wetter longer and more easily accessed upon wetting and later abandoned upon drying.

Streams in the Great Plains region of the United States represent particularly harsh environments with rapid transitions between flooding and drying [Dodds et al. 2004]. Fishes within these temporary streams are adapted to a harsh hydrologic flow regime and can migrate to permanent waters, reproduce rapidly, and persist in isolated pools with poor water quality [Pauloumpis 1958; Labbe and Fausch 2000]. Research has shown that fishes in isolated pools are able to withstand low dissolved oxygen concentrations and elevated water temperatures up to nearly 40 deg C [Erman and Leidy 1975; Mundahl 1990]. Fishes may also survive drought conditions by migrating downstream to perennial reaches [Harrel et al. 1967]. Fishes surviving in perennial stream reaches and/or pools readily re-colonize temporary reaches when surface water connectivity resumes [Reeves 1979].

Amphibians

As their name implies, amphibians (*amphíbios [Greek]: amphi = both, bio = life*) inhabit both land and water. Stream reaches alternating between wet and dry states can therefore serve as ideal habitat for animals adapted to a combination of terrestrial and aquatic life. Temporary headwaters may provide ideal habitat for salamander breeding where episodic flow results in the lack of significant fish populations and, in turn, decreased resource competition and predation pressure [Wilkins and Peterson 2000]. Supporting this, spring salamanders (*Gyrinophilus porphyriticus*) (Figure 5A) and two-lined salamanders (*Eurycea bislineata*) (Figure 5B) have been found in greater abundances in fishless headwater, albeit perennial, streams than in perennial streams with predatory fishes [Barr and Babbitt 2002; Lowe and Bolger 2002]. Wilkins and Peterson [2000] found fishless, non-channelized spring-fed seeps in the Pacific Northwestern United States supported both larval and adult Columbia torrent salamanders (*Rhyacotriton kezeri*) (Figure 5C). In a study of over 30 perennial and intermittent streams in northern California (USA), coastal giant salamanders (*Dicamptodon tenebrosus*) and black salamanders (*Aneides flavipunctatus*) were found to be significantly more abundant along intermittent stream reaches which were shaded, damp, and hydrologically disconnected from perennial sites [Welsh et al. 2005]. Collectively, these conditions result in ideal sites for egg deposition and predator avoidance.

Figure 5. Examples of salamander species that may be found in either temporary streams or fishless perennial headwaters. A) Spring salamander (*Gyrinophilus porphyriticus*) [Photo: J. Butler] B) Northern two-lined salamander (*Eurycea bislineata*) [Photo: M. Jennette] C) Columbia torrent salamander (*Rhyacotriton kezeri*) [Photo: M. Leppin].

Anurans, like salamanders, tend to inhabit temporary stream reaches for both predator avoidance and dietary resources. In an intermittent stream in west-central Kentucky (USA), pickerel frog (*Rana palustris*) and American toad (*Bufo americanus*) females were found to selectively oviposit in areas where few if any fishes were present [Holomuzki 1995]. Moreover, tadpoles of both species reduced their activity in the presence of fishes by detecting fish chemical cues and preferentially inhabited areas of the stream most inaccessible to swimming fishes (e.g., channel margins and isolated pools) [Holomuzki 1995]. Inger and Colwell [1977] compared distributions of anuran species in perennial and temporary forested streams in northeastern Thailand and noted that amphibians inhabited temporary sites in significantly greater numbers relative to permanent reaches. They also found that certain frog species (e.g., *Rana nigrovittata* and *Rana pileata*) were essentially confined to small intermittent streams [Inger and Colwell 1977]. In a study of algal quality as a dietary resource in perennial and temporary Australian streams, Peterson and Boulton [1999] found that tadpoles were better able to digest algae from newly rewetted temporary reaches, suggesting a higher quality of algal dietary resources in temporary versus permanent waters.

Streamside Mammals, Birds, and Reptiles

In addition to temporary channels themselves, riparian zones along ephemeral and intermittent streams may provide habitat and dietary resources for mammals, birds, and reptiles. When temporary streams dry, isolated pools often result. Fragmentation of stream reaches into pools effectively traps and increases the density of aquatic organisms including invertebrates, amphibians, and fishes [Boulton and Lake 1992]. As a result, pools become dietary hot spots where food may be obtained at minimal energy cost by riparian mammals, birds, and snakes [Metzger 1955; Tramer 1977; Kephart 1982; Dowd and Flake 1985].

The corridors of temporary streams may also serve as preferential edge habitat for a number of species. In the xeric southwestern United States, for instance, it is estimated that 80% of all animals use temporary stream riparian habitat and/or dietary resources at some life stage, and that greater than 50% of breeding bird species nest primarily in temporary stream riparian habitats [Krueper 1993].

Seidman and Zabel [2001] found that intermittent portions of a California (USA) stream network supported similar amounts of bat activity as perennial reaches. Similarly, Ozark big-eared bats (*Plecotus townsendii ingens*) were found to forage heavily along intermittent streams in Oklahoma (USA) [Clark et al. 1993]. Sonoran mud turtles (*Kinosternon sonoriense*) were found to be thriving alongside intermittent mountain stream corridors in New Mexico (USA) [Stone 2001]. Birds, including red-tailed hawks (*Buteo jamaicensis*) and Gila woodpeckers (*Melanerpes uropygialis*), depend on ephemeral stream riparia in arid Arizona (USA) for nesting sites found only in these terrestrial-aquatic interfaces [Johnson and Lowe 1985].

PLANT HABITAT

As well as providing valuable wildlife habitat, temporary streams and their riparia harbor substantial floral communities, particularly in arid or semi-arid regions. Streambed and streamside vegetation plays a significant role in temporary stream ecosystem structure and function. Within-channel and riparian vegetation which are both heavily influenced by flow regime [Poff et al. 1997], provide channel and bank roughness, buffer high flows, stabilize banks, mitigate wind and water erosion, and trap particulates [Levick et al. 2008]. Because temporary streams generally have a larger channel edge-to-width ratio than perennial channels, the proportion of the streambed and riparian zone that can be colonized by vegetation is often greater in temporary reaches compared to those that flow year-round [Fritz et al. 2006].

In semi-arid and arid regions, temporary streams and their riparia are often the only places in the watershed with soil moisture levels necessary to support a substantial plant community. As such, ephemeral and intermittent stream corridors may be hot spots of plant diversity and abundance relative to their watersheds [Warren and Anderson 1985]. Studying summer dry streams in northern California (USA), Waters et al. [2001] found that the mean number of plant species in the herbaceous layer along channels as narrow as 0.9 to 1.3m was significantly greater than the mean in upland sites. Temporary stream channels themselves can also be zones where terrestrial plants can establish. McBride and Strahan [1984] found that woody species including willow (*Salix* sp.), Fremont cottonwood (*Populus fremontii*), and mule fat (*Baccharis viminea*) readily established in gravel bars in a seasonally intermittent stream in northern California (USA). However, flow regime extremes of summer drought and channel-scouring winter floods led to plant mortality and loss on the gravel bars [McBride and Strahan 1984].

Temporary streams and riparia may also maintain diverse soil seed banks and support unique plant species and high plant diversity. The seed banks of temporary streams in arid regions may support wetland plant communities during portions of the year when moisture levels are sufficient for hydric species [Brock and Rogers 1998; Goodson et al. 2002; Stromberg et al. 2005; Capon 2007; Stromberg et al. 2009]. Studying the species composition of perennial and temporary stream seed banks in the Hassayampa River (Sonoran Desert, Arizona, USA), Stromberg et al. [2009] found that an ephemeral site nearly 50km from the closest downstream perennial reach harbored hydric seeds. Stromberg et al. [2009] also found that some xeric species were endemic to ephemeral but not perennial portions of the Hassayampa river network. In a study of the spatially intermittent Cienega Creek (Sonoran Desert, Arizona, USA), Stromberg et al. [2009b] found that during wetter portions of the year, ephemeral reaches boasted vegetative species richness levels equal to and sometimes greater than those in perennial reaches. But, a similar Sonoran Desert study found that as stream flows became more intermittent, diversity and cover of herbaceous species along the channel declined [Stromberg et al. 2007]. Studying moss and liverwort distributions in forested headwater stream networks throughout the United States, Fritz et al. [2009] found a general pattern of greater species bryophyte richness in temporary streams relative to those that flowed year-round.

As flow regime shifts from perennial to temporary, vegetation composition shifts toward increasingly drought-tolerant species, vegetative cover declines, trees give way to shrubs, and canopy height and cover decline [Leenhouts et al. 2006; Stromberg et al. 2007]. Working in temporary streams in the Sonoran Desert (Arizona, USA), Stromberg et al. [2005] studied the response of streamside herbaceous vegetation to changes in stream flow permanence. They found that streamside herbaceous cover and species richness declined continuously across gradients of flow permanence during the early summer dry season, and that composition shifted from hydric to mesic species at sites with more intermittent flow [Stromberg et al. 2005].

Biogeochemical Cycling

The biogeochemical functions of temporary streams include cycling of elements and compounds, particle retention, and organic matter transformation and transport [Levick et al. 2008]. Biogeochemical cycling occurs primarily through chemical transformations mediated by redox potentials. Reduction and oxidation reactions are governed by the soil profile, wind, and hydrology [Brinson et al. 1995]. As active zones of cyclical wetting and drying, temporary stream sediments are marked by alternating anoxic and oxic periods and, in turn, alternating reducing and oxidizing conditions. The pattern of temporary stream intermittency (i.e., the timing, duration, and frequency of surface flow) is likely to govern many in-stream processes, particularly biogeochemical rates.

Biogeochemical "hot spots" are areas that show disproportionately high reaction rates relative to the surrounding matrix. "Hot moments" are short periods of time that show disproportionately high reaction rates relative to longer intervening time periods [McClain et al. 2003]. Wetting-drying cycles create hot spots and hot moments [McClain et al. 2003] and have been shown to increase nitrate loss in soils [Patrick and Wyatt 1964; Reddy and Patrick 1975; Tanner et al. 1999; Eaton 2001; Venterink et al. 2002] via a coupled nitrification-denitrification process. Aerobic soils present during dry conditions promote nitrification of ammonia [Qiu and McComb 1996]. Under saturated conditions, soils become anaerobic, nitrate delivery to sediment microbial communities is increased, and the nitrate substrate is reduced via denitrification [Holmes et al. 1996]. Baldwin and Mitchell [2000] have observed this dry–wet / nitrification–denitrification coupling in river floodplains, and Pinay et al. [2007] observed high denitrification rates in a European floodplain rewetted via rainfall and flooding following a dry period. Given their cyclic dry–wet nature, temporary stream sediments are likely to function similarly to episodically inundated floodplains and prove to be biogeochemical hot spots undergoing hot moments where microbial respiration and denitrification rates are enhanced. While literature is sparse on how patterns of stream intermittency govern such rates, increased denitrification rates have been observed during high moisture conditions in temporary streams [Fisher et al. 2001; Rassam et al. 2006].

Re-wetting of dry soil can kill up to 50% of that soil's microbial biomass via osmolysis [Kieft et al. 1987], in turn resulting in release of carbon and nutrients [Marumoto et al. 1982]. Shortly after this initial loss of soil microbial biomass, water, carbon, and nutrient availability may then stimulate rapid increases in microbial biomass [Kieft et al. 1987] and microbial processing (e.g., N mineralization, nitrification, denitrification) [Davidson et al. 1990; Fisher and Whitford 1995].

Temporary streams may also function as zones of carbon storage and processing [Towns 1985; Dieterich and Anderson 1998; Halwas and Church 2002; Acuna et al. 2004]. At the most distal branches of river networks, headwater temporary streams have a small average width and, in turn, significant canopy cover and subsequent allochthonous organic matter loading [*sensu* Vannote et al. 1980]. Shallow water, low stream power, and a generally high number of in-channel retentive structures can lead to increased sediment retention in temporary reaches [Dieterich and Anderson 1998]. Sediment retention can in turn trap organic matter via burial [Brinson et al. 1995], particularly in channels dominated by fine sediments [Herbst 1980; Metzler and Smock 1990]. Furthermore, organic matter decomposition rates within temporary streams are often slow relative to breakdown rates in perennial streams [Tate and Gurtz 1986; Fritz et al. 2006]. Slow rates of decomposition are due in part to periods of desiccation [Tate and Gurtz 1986], low microbial growth rates during dry conditions [Witkamp and van der Drift 1961], negligible or no physical breakdown of organic matter under low or no flow conditions, and reduced macroinvertebrate shredder densities [Kirby et al. 1983; but see Hill et al. 1988]. Significant allochthonous loading coupled with high retention and low decomposition rates result in the buildup of in-stream organic matter. Dry seasons or drought conditions have resulted in significant organic matter buildup in temporary streams [Larned 2000; Acuna et al. 2004].

Large standing stocks of benthic organic matter (BOM) may fuel heterotrophic activity when channels rewet [von Schiller et al. 2008]. Organic material stored in small streams can be broken down and transformed into forms more bioavailable to biota in perennial downstream waters [Richardson et al. 2005]. Although coarse particulate organic matter

(CPOM) may accumulate during dry periods, temporary streams characterized by seasonal flow may actively flush CPOM standing stocks downstream, especially if flow resumes rapidly [Gurtz et al. 1988; Hill et al. 1992; Acuna et al. 2004]. In addition to flood-induced flushing, organic matter may be exported from temporary streams via leaching or wind- or baseflow-mediated displacement [Brinson et al. 1995].

Connectivity

Despite temporal and/or spatial discontinuities in surface flow, ephemeral and intermittent streams are intimately linked hydrologically and ecologically to their watersheds and to perennial waters [Cummins and Wilzbach 2005; Alexander et al. 2007; Nadeau and Rains 2007]. Larned et al. [2010] define *hydrologic connectivity* as "the presence or absence of flowpaths between persistent patches of aquatic habitat" while Freeman et al. [2007] expand the definition, referring to it as "the water-mediated transport of matter, energy, and organisms within or between elements of the hydrologic cycle." Although only connected episodically via surface flow to perennial waters, temporary streams, like perennial flowing waters, can move substantial amounts of water, nutrients, sediments, and animal and plant propagules throughout watersheds and waterways. Connections between temporary and permanent or other temporary waters can be longitudinal (channel ←→ channel), lateral (channel ←→ floodplain), vertical (channel ←→ groundwater), and temporal (across time) [Ward 1989; Freeman et al. 2007] (Figure 6).

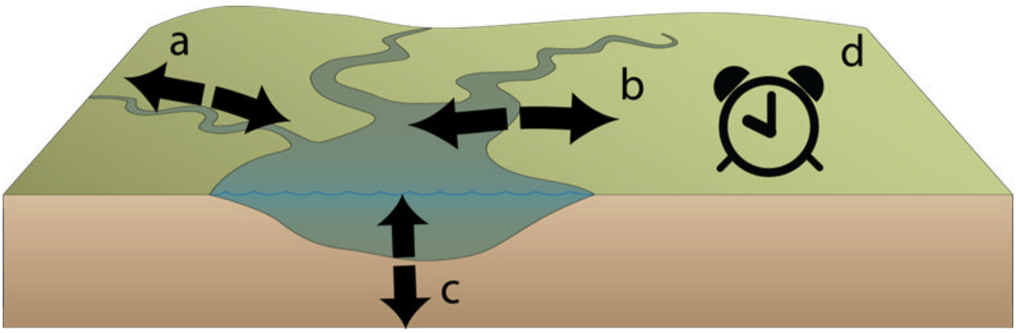

Figure 6. Four dimensions of connectivity within lotic ecosystems (after Ward 1989). a) longitudinal connectivity (channel ←→ channel). b) lateral connectivity (channel ←→ floodplain). c) vertical connectivity (channel ←→ groundwater). d) temporal connectivity (across time). [Modified from symbols courtesy of the Integration and Application Network (ian.umces.edu/symbols/), University of Maryland Center for Environmental Science].

Temporary streams act as carbon delivery pathways critical to the ecological integrity of river networks. Carbon is an important energy source for aquatic organisms, forming the base of food webs in lotic networks. Considering their abundance and periodic connectivity to permanent downstream reaches, Fritz et al. [2006] hypothesize that temporary headwaters are major downstream contributors of organic matter. Moreover, research has shown that temporary waters, including streams and floodplains, although only episodically connected to the larger river network, can be significant sources of organic carbon to rivers. Studying anabranching channels (sections of a river or stream that divert from the main channel or stem and rejoin the main stem downstream) episodically connected to the Macintyre River (southeastern Australia), Thoms [2003] and Thoms et al. [2005] found that these temporarily connected reaches not only make up as much as 87% of the river length, but that they are significant contributors of dissolved organic carbon (DOC) to the river mainstem. The amount and timing of organic carbon exported from temporary streams is a function of the pattern of hydrologic connectivity, streambed hydraulic conductivity, soil organic matter, benthic organic matter and woody debris, rate and state of organic matter decomposition, and allochthonous and autochthonous carbon inputs [Lee et al. 2004].

Ephemeral and intermittent streams may also transport nutrients throughout river networks. Alexander et al. [2007] quantified water and nitrogen transport from headwater streams to downstream waters. They found that headwaters, including ephemeral and intermittent tributaries, contributed approximately 70% of the mean annual water volume and 65% of the nitrogen flux to second order streams [Alexander et al. 2007]. Additional research on the role of temporary streams on downstream nutrient loading, however, is sparse.

Temporary streams may also serve as connectivity corridors for both animal and plant species. Surface flow in ephemeral and intermittent streams enables movement of obligate aquatic animals (e.g., fishes, invertebrates lacking a terrestrial and/or flying adult stage) and *hydrochorous* plants (those with seeds that disperse via water) into otherwise disconnected waters. Moreover, the riparian corridors along temporary streams can serve as migratory pathways for a variety of animal species. Dispersal throughout temporarily flowing portions of river networks allows for genetic exchange between subpopulations that are isolated for some portion of the year as well as opportunities for recolonization of periodically disconnected and/or uninhabitable reaches or pools [Levick et al. 2008]. Dispersal between temporarily connected waters may also allow for geneticexchange between otherwise isolated subpopulations, thereby enhancingmetapopulation genetic diversity and persistence. Some plant species exist along river networks as metapopulations [Menges 1990], with flow-mediated seed dispersal helping to maintain subpopulation connectivity. Meyer et al. [2007] warn that the loss of connectivity between small headwaters – including ephemeral and intermittent reaches – and larger downstream waters will detrimentally impact the biodiversity not only of the headwaters themselves, but of the entire river network.

Even in the absence of surface hydrologic connectivity, ephemeral and intermittent streams can contribute water, carbon, and nutrients to perennial streams. Surface water in temporary streams can be transferred to groundwater reserves or hyporheic flowpaths [Fisher and Grimm 1985; Belnap et al. 2005; Izbicki 2007]. This subsurface water may reemerge downstream in perennial waters or springs where it can be an important source of baseflow, energy, and nutrients [Fisher and Grimm 1985].

ANTHROPOGENIC DISTURBANCES

Because of their small size, large edge-to-width ratio, and intimate linkage to the catchments they drain, temporary streams are likely to be more sensitive to disturbance than larger perennial streams [Bull 1997]. Adding to their risk is the fact that temporary streams are often unmarked on standard topographic maps and have been "historically neglected" by ecologists [Larned et al. 2010]. And, perhaps not surprisingly, temporary streams often receive less regulatory protection than perennial reaches, and less mitigation may be required for their degradation [Johnson et al. 2009]. In fact, a recent Supreme Court case in the United States ruled that streams must either be "relatively permanent, standing or flowing" or significantly impact the biological, chemical, or physical integrity of perennial waters in order to be protected from dredge and fill under the US Clean Water Act [*Rapanos v. United States* 2006]. As a result of all these factors, temporary streams represent particularly vulnerable ecosystems.

Direct Disturbance

Direct anthropogenic disturbance to temporary streams may result from numerous activities including water abstraction, livestock grazing, land clearing, timber harvesting, flow diversion, agriculture, road construction, channelization and loss of riparian vegetation / floodplain connectivity, damming, urbanization, mountaintop mining, and even burial. These disturbances tend to cause alteration of the natural flow regime, loss of faunal and floral habitat, impaired water quality, and/or physical channel / floodplain modification [Dodds et al. 2004]. Many intermittent prairie streams in the Great Plains (central USA) that once meandered through native grasslands have been anthropogenically straightened to more efficiently drain extensive cropland or urban areas [Dodds et al. 2004]. The same is likely also true for once naturally braided channels in the Coastal Plain region of the eastern United States [Walter and Merritts 2008]. The result of these actions is an altered hydrology from seasonally intermittent to more ephemeral and an increase in sediment, nutrient, and contaminant export relative to native prairie streams [Dodds et al. 2004].

Furthermore, increased groundwater abstraction and flow diversion for municipal, agricultural, and industrial use can lead to excessive stream drying, loss of connectivity corridors, and elimination of pool refugia for animals [Uys and O'Keeffe 1997; Webb and Leake 2006]. Groundwater abstraction from the Ogallala High Plains aquifer in the Great Plains (central USA) has led to a lowering of the water table and resultant intermittency in streams that until recently were perennial [Dodds et al. 2004]. Impoundments including dams and farm ponds along temporary stream reaches disrupt connectivity of the lotic system and migratory and dispersal pathways for both plants and animals [Fausch and Bestgen 1997; Dodds et al. 2004].

Urbanization and mountaintop mining, in particular, may also lead to ephemeral, intermittent, and small headwater stream loss – often via direct burial. Elmore and Kaushal [2008] investigated the impact of urbanization on stream loss and found that 70% of streams with drainage basins < 260 ha (1 mi^2) within Baltimore City, Maryland (USA) were lost due to burial. Modeling stream length and hydrologic permanence in Hamilton County, Ohio (USA), Roy et al. [2009] reported 93% and 46% county-wide decreases in ephemeral and intermittent channel length, respectively, in urban versus forested catchments. Mountaintop mining may also result in temporary stream burial [Palmer et al. 2010]. In the central Appalachian ecoregion of the United States, mountaintops are commonly removed to access buried coal. Excess rock, or mine "spoil," is pushed into nearby valleys where it buries existing streams. Particularly vulnerable to these valley fills are small, temporary headwaters.

Indirect Disturbance

Climate change is perhaps the most significant indirect anthropogenic disturbance facing temporary streams. Because temporary stream hydrology is tightly linked to patterns of temperature and precipitation, these waters are particularly sensitive to climatic changes. Both the frequency and intensity of drought and, in turn, stream drying are predicted to increase under current climate change scenarios [Lake et al. 2000; Palmer et al. 2008; Brooks 2009] (Figure 7). Moreover, climate change models predict more variable temperature and precipitation patterns that will lead to increased frequency of flow extremes (e.g., flooding and drying) [Lake et al. 2000], thereby fundamentally altering natural flow regimes in temporary streams and, in turn, stream structure and function [*sensu* Poff et al. 1997]. Such changes will be exacerbated in urban areas where flow variability is already enhanced compared to forested regions [Nelson et al. 2008]. Schindler [1997, 2001] suggests that global climate change will cause increased evapotranspiration in much of North America, in turn resulting in increased temporary stream occurrence, particularly among headwaters. Refined modeling and forecasting efforts aid in the prediction of climate change impacts on temporary waters and should help guide proactive management plans. Yet, long-term monitoring will be necessary to accurately document the impacts on temporary stream hydrology and ecology [Conly and van der Kamp 2001].

Figure 7. Headwater stream reach near the Speed River in southern Ontario, Canada. A) Stream with surface water present under average autumn climatic conditions (2008). B) Stream lacking surface water during an excessively dry autumn (2007). [Photos: C. Febria]

Human-induced intermittency, both direct and indirect, will have clear and significant impacts on the ecology of stream networks [Brooks 2009]. Increased intermittency and fragmentation of temporary waters can lead to fishery declines, loss of migratory pathways and ecosystem connectivity, disrupted downstream flow regimes, loss of biogeochemical processing capacities, and degradation of the ecological integrity of stream networks as a whole [Larned et al. 2010]. Lack of connectivity between once linked reaches or pools can lead to population bottlenecks within species unable to encounter and reproduce with conspecifics [Labbe and Fausch 2000]. Maintaining connectivity within temporary stream networks is critical for the conservation of populations and biodiversity [Labbe and Fausch 2000]. Direct anthropogenic induced intermittency resulting in rapid transitions from permanent flow regimes to temporary surface flow patterns is common and becoming more frequent [Fu et al. 2004; Bernard and Moetapele 2005; Qi and Luo 2005; Hao et al. 2008].

Indirect human-induced intermittency resulting from climate change will occur gradually and in line with broad drying patterns [Larned et al. 2010].

CONCLUSION

Temporary streams have a unique hydrologic flow regime that places them at an interface between land and water. Although often poorly mapped, recognized, and protected, they are abundant, ubiquitous, and critical to the ecological health of lotic networks. Collectively, temporary streams provide invaluable animal and plant habitat, hot spots for biogeochemical processing, and corridors of hydrologic and ecologic connectivity throughout river systems. Yet, they are faced with a multitude of direct and indirect anthropogenic disturbances to their hydrology, ecology, and even existence.

Temporary streams have been historically neglected [Larned et al. 2010] – not only with respect to scientific study – but more critically as ecosystems vital to the physical, chemical, and biological integrity of entire river networks. The functions of temporary streams must be recognized and valued in order to map, manage, and protect them properly [Levick et al. 2008]. Yet management and protection of ephemeral and intermittent streams - and arguably all small streams – is hindered by the lack of viable assessment methods and reasonable ecological expectations [Fritz et al. 2008]. There exists, therefore, a need for methods to scientifically study the structure and function of temporary streams. Considering their abundance, studies of the cumulative impacts of temporary streams and their loss on the chemical, physical, and biological integrity of lotic networks are also needed.

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Chapter 8

EVALUATION OF NON-POINT SULFATE CONTAMINATION IN RIVER BASINS

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ABSTRACT

Sulfuric acid is discharged from acid sulfate soils or mining sites in the watershed of rivers, and this sulfuric acid affects aquatic communities. In basins with coal mining sites, sulfate migrates into river systems via mine drainage as well as via ground water flow.

In order to evaluate the non-point contamination of sulfate via ground water into river ecosystems, we analyzed the chemical properties of river water from the upper-most basins to the river mouth at 1-2 km intervals.

We surveyed two rivers in the northern part of Kyushu Island, Japan. The Ongagawa River basin has many abandoned coal mining sites and the SO_4^{2-} /Cl^- concentration ratio (w/w) reached up to 7.0 in the middle basin, whereas the ratio in the Chikugogawa River with few coal mining sites in the basin was constantly 1.0-2.0 in the middle basin, in contrast to the extremely high value (20.0) at the upper-most basin of the river because of the effect of sulfur emissions from active volcanoes. Thus, we found that the SO_4^{2-} /Cl^- concentration ratio of river water with abandoned coal mining sites in its basin was much higher than was the case with a river without coal mining sites in the basin. The SO_4^{2-} /Cl^- concentration ratio reached a higher value in the basin of a tropical peat swamp forest with acid sulfate soils in central Kalimantan, Indonesia, and the ratio can be used as a useful index for the qualitative evaluation of the non-point contamination of sulfates in river systems.

Keywords: acid sulfate soil, Chikugogawa River, coal mining, non-point pollution, Ongagawa River, river basin.

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INTRODUCTION

Pyrite is distributed widely in the sediments and bedrocks around the world. One of the serious environmental problems caused by pyrite oxidation is the formation of acid mine drainage (AMD) from mine site spoils. Mining operations produce large amounts of waste tailings, which are usually deposited in open-air impoundments. Waste tailings containing metal sulfates such as pyrite lead to the production of acid rock drainage, which contaminates the environment with heavy metals and sulfuric acid. Many reports describe the effects of sulfuric acid from mine wastes on vegetation and ecosystems (e.g., Meyer et al. 1999, Bachmann et al. 2001, and Werner et al. 2001). Acid mine drainage containing a high concentration of sulfuric acid causes aquifer pollution with a high groundwater sulfate concentration, a low pH and enhanced heavy metal contents, e.g., Ni, Co, Cu, Pb, As or Zn (Andersen et al. 2001).

In former open-cut brown coal mining areas dating from the early 20th century, for example in The Lower Lusatian lignite mining district in Germany, there still remain many open mining casting lakes with high concentrations of sulfuric acid and extremely high acidity (pH = 1.5 - 2.5). Sulfuric acid derives from the mine spoils surrounding the lakes, and the continuous supply of sulfuric acid into the lake accelerates the acidification of the lake water. Furthermore, at this point most of the areas have not been reclaimed. An advanced method of open-cut mining of brown coal using a conveyer bridge dump can prevent the pyrite oxidation by maintaining the stratigraphy of the Tertiary strata to the greatest degree possible (Wisotzky and Obermann 2001). However, the rehabilitation of vegetation after open mining in such areas is difficult because of the contamination by pyrite of the topsoil.

Pyrite is deposited primarily in marine systems, and hence environmental problems caused by pyrite oxidation after agricultural peatland development usually appear in coastal peat swamps. Hall et al. (2006) investigated the distribution of sulfidic sediments in inland wetlands within a river basin and evaluated the risk of sulfuric acid as well as heavy metal contamination in that river basin after water table drawdown in the wetlands. They demonstrated that sulfidic sediments appear in some mires and exposure of sediments to the atmosphere can lead to the oxidation of pyritic minerals and the production of acid.

Accumulated sulfuric acid after pyrite oxidation causes acidification not only of the soil but also of the surrounding environments. Discharged sulfuric acid from the soil causes the acidification of river water and subsequent effects on the littoral zone. He et al. (1997) showed that acid mine drainage strongly affects acidity and heavy metals in river water in a basin with mining sites. Johnson and Thornton (1987) showed that Cu, Zn and As concentrations of river water that is affected by acid mine drainage increased in the high-precipitation season (winter), implying that these contaminants discharged into the river via surface drainage. Olías et al. (2004) demonstrated that sulfate ion as well as heavy metals concentrations of river water exhibited maximum values at the first rainfall event after dry season. The first rainfall after dry season provoked redissolution of sulfate salts that had precipitated during the dry season due to the intense weathering of the pyrite.

There are many abandoned coal mining sites in the northern part of Kyushu Island, Japan, and the contamination of acids and heavy metals has been evident in some parts of the river basin. Mine drainage water discharges from the open wellhead as well as through ground water flow, and the effect of discharged water via ground water flow is not easily evaluated.

In order to estimate the effects of mine drainage discharged via the ground water system on a river basin, we monitored the river water chemistry in a basin with high spatial resolution. We monitored the river water of the Ongagawa River, which has several former coal mining sites within its basin, at ca. 1-2 km intervals, and as a comparative reference we monitored the river water of the Chikugogawa River, which has few coal mining sites within its basin. Then we evaluated the effects of coal mine drainage from the abandoned coal mining sites on the freshwater systems, including effects via ground water with reference to sulfate contamination.

The method used in this analysis has been used for the evaluation of sulfuric acid contamination in the tropical peat swamp forest area in Central Kalimantan, Indonesia (Haraguchi 2007; Haraguchi et al. 2007) after a land use change to agricultural purposes and the consequent peat decomposition. It is evident that the pH of the bottom layer of the coastal peat in the tropical peat swamp forest maintained a lower pH value compared to that for the peat cores in the upper basin of the river (Haraguchi et al. 2000; Haraguchi et al. 2005). Sulfuric acid produced from pyrite oxidation was able to lower the pH of the bottom layer of the peat to a value lower than that of the bottom peat layer in the upper basin of the river (Haraguchi et al. 2005). The concentration of SO_4^{2-} in the peat pore water at the bottom layer of peat in the coastal peat is much higher than that for the peat in the upper basin (Haraguchi et al. 2005). This implies that the bottom layer of the coastal peatlands is strongly affected by the sulfuric acid originating from pyrite oxidation within the mineral sediment under the peat layer. Effects of sulfuric acid discharge from non-point sources on the freshwater system have been evaluated in this tropical peat swamp forest that is strongly affected by sulfuric acid contamination after pyrite oxidation by measuring river water chemistry at 1-3 km intervals within the basin. Using the same method used in the tropical peat swamp forest, we investigated the effects of non-point sulfate sources from abandoned coal mining sites on river ecosystems in northern Kyushu.

METHODS

Study Sites

The Ongagawa River (origin of the main stream: 33° 28' 28" N, 130° 48' 30" E, 978m a.s.l.; river mouth: 33° 54' 00" N, 130° 39' 42" E) is located in the northern part of Kyushu Island, Japan (Figure 1). The total length of the main stream of the Ongagawa River is 61 km, and the total area of the basin including all tributaries is 1,026 km^2. The largest tributary is the Hikosangawa River (origin: 33° 28' 03" N, 130° 55' 03" E, 742m a.s.l.) and it connects to the main stream of the Ongagawa River at a point 19 km upstream from the mouth of the river. The Ongagawa River and the Hikosangawa River originate from a mountainous area with a forested basin, and run through an agricultural land system mainly used as paddy fields, and then flow within the urbanized area in the lower basin. The averaged annual precipitation of Kama in the upper most basin of the river (33˚ 28' 32.65" N, 130˚ 48' 39.67" E, 502 m a.s.l.) is 2,355.8 mm (data from the Japan Meteorological Agency between 1971 and 2000). About 36 % of the annual precipitation was within June and July.

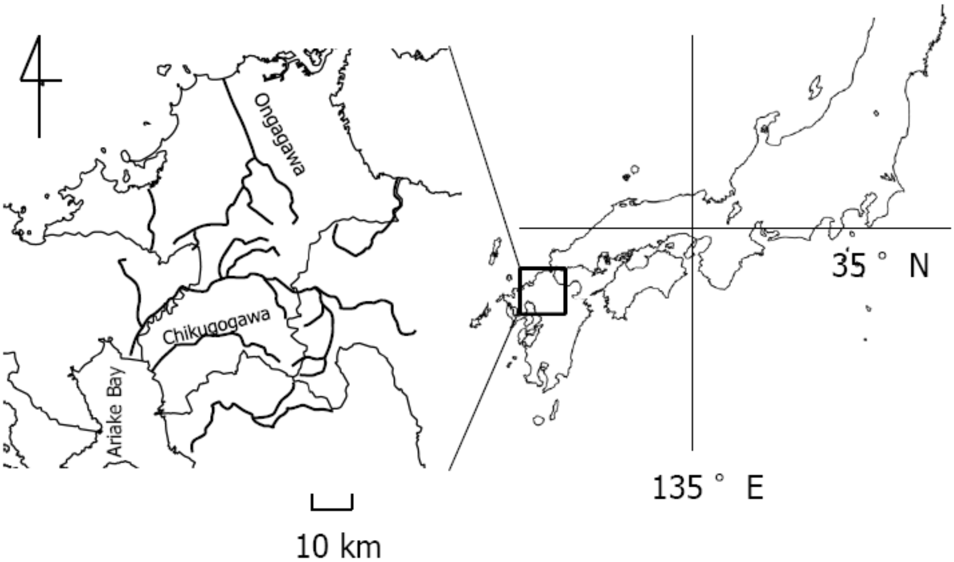

Figure 1. Map showing the area studied in northern Kyushu, Japan, and the investigated rivers: Ongagawa River and Chikugogawa River.

The Chikugogawa River (origin of the main stream: 33° 05' 10" N, 131° 11' 57" E, 1140m a.s.l.; river mouth: 33° 05' 06" N, 130° 24' 37" E) is located in the north-western part of Kyushu Island (Figure 1). The total length of the main stream is 143 km, and the river basin area is 2860 km^2. The largest tributary is the Kusugawa River (origin: 33° 05' 55" N, 131° 15' 49" E, 1256m a.s.l.) and it connects to the main stream of the Chikugogawa River at a point 77.5 km from the river mouth. The origin of the Chikugogawa River is within the Aso-Kuju volcanic mountain area, and the land use of the middle basin of the river is predominantly paddy fields and orchards. The river then flows through an urbanized area in the lower basin of the river. Areas downstream from the floodgate of the river (Chikugo-oozeki located 24.5 km from the river mouth) are affected by sea water inundation. Averaged annual precipitation in Bougatsuru, the uppermost basin of the Kush River (33˚ 05' 36.34" N, 131˚ 15' 50.34" E; 1240 m a.s.l.), is 2724.6 mm (data from the Japan Meteorological Agency between 1971 and 2000). About 36 % of the annual precipitation occurs within June and July.

Analytical Methodology

In the present study, we investigated the spatial variation in concentrations in river water of the Ongagawa River and the Chikugogawa River, in order to clarify the spatial distribution of sulfate contamination along the entire river basin.

In the Ongagawa River, surface river water was sampled at 83 points (including a tributary, the Hikosangawa River) at 1 - 2 km intervals every month from May 2002 to May 2006. In the present analysis, we used the data from April 2005 to March 2006 ranging from the river mouth to the uppermost basin of the Ongagawa River (50 sampling points), which was obtained from the data base of Ongagawa River water chemistry.

In the Chikugogawa River, we investigated surface river water chemistry at 157 sampling points ranging from the river mouth up to 139km upstream (origin of a tributary, the Kusu

River). Water samples were collected at ca. 1 km intervals from the Chikugogawa River three times on 22 August, 13 November and 17 December 2008.

We collected 100 mL of surface water at each sampling point using a ladle from the riverside or with a water sampler from bridges. Samples were transported to the laboratory within one day and kept at 5 ˚C. The electrical conductivity (EC) and pH values were measured within 24 hours after sampling and before filtration. Within 24 hours after sampling, approximately a 10 mL sample was filtered with a cellulose acetate membrane filter (MILLIPORE, 0.22 μm) and kept at 5 ˚C before measurement of major ion concentrations. We measured major cations (NH_4^+, Na^+, K^+, Mg^{2+}, Ca^{2+}) and anions (NO_2^-, NO_3^-, Cl-, SO_4^{2-}, PO_4^{3-}) with an ion chromatograph (DX-120, Dionex). Among measured parameters, we used Cl^- and SO_4^{2-} for the present analysis of sulfate contamination in the river basin.

Existing Data for the Rivers in a Tropical Peat Swamp Forest

Data for the rivers in a tropical peat swamp forest in Central Kalimantan, Indonesia, were used as a reference point for the rivers in northern Kyushu (Haraguchi 2007; Haraguchi et al. 2007). Water chemistry of two rivers in a tropical peat swamp forest, the Sebangau River and the Kahayan River, was surveyed in September 2003, 2004 (dry season) and March 2004, 2005 (rainy season). The Sebangau River originates in a peat swamp forest to the west of Palangkaraya City. Whole catchments of the Sebangau River are covered by peatlands, and hence the river water contains high concentrations of humic substances (dark brown color) and background pH is 3.5-4.0. Six main tributaries flow into the main stream of the Sebangau River. Six main canals have been connected to the Sebangau River from the eastern part of the main stream. Water samples were collected from the river mouth to Kya (uppermost part of the river at 177 km from the river mouth), including points in the Paduran canal (artificial) and four natural tributaries (the Paduran, Bangah, Rasau, and Bakung Rivers).

The Kahayan River originates at Kahukung Mountain and flows east of Palangkaraya City (224 km from the river mouth). Most of the peat land is distributed downstream of the Kahayan River from Palangkaraya City, and hence concentrations of humic substances in the river water are much lower than those found in the Sebangau River, and pH of the water around Palangkaraya City is 5.5-7.0, higher than in the Sebangau River. Five main canals from the western part and also five canals from the eastern part have been connected to the main stream of the Kahayan River downstream from Palangkaraya. The Rungan River merges 235 km upstream from the river mouth. Data for the Kahayan River and the Rungan River were obtained up to 326 km and 333 km from the river mouth, respectively, but this paper presents only the data for the Kahayan River from the river mouth to326 km from the river mouth including four artificial canals (Pangkoh, Kanamit, Basarang, and Pulangpisau). Data from the river mouth to the 45.1 km point as well as that for Kanamit Canal are missing for September 2003.

Water samples were collected at the center of each river at intervals of every 1.0-3.0 km along the rivers. Each water sample was directly collected from a boat by using a ca. 1,000 ml plastic tub at each sampling point. Water temperature, pH and EC (electrical conductivity) of the collected water were measured just after the water sampling by using a portable pH meter (D-25, Horiba Co. Ltd., Kyoto, Japan) and EC meter (ES-12, Horiba Co. Ltd., Kyoto, Japan).

Water samples were filtered within 12 hours after sampling using a 0.45 μm cellulose acetate membrane filter (Advantec Co. Ltd., Tokyo, Japan) and stored in 2.0 ml plastic tubes at room temperature before chemical analysis. Major cations (NH_4^+, Na^+, K^+, Mg^{2+}, Ca^{2+}) and anions (NO_2^-, NO_3^-, Cl-, SO_4^{2-}, PO_4^{3-}) were determined using an ion chromatograph (Dionex Model DX-120, Japan Dionex Co. Ltd., Tokyo, Japan). Among measured parameters, we used Cl^- and SO_4^{2-} for the analysis of sulfate contamination in the river water.

RESULTS AND DISCUSSION

The ratio of SO_4^{2-}/Cl^- of the river surface water of the Ongagawa River registered constantly in the range 1.0-1.5 in the upper most basin (40-58 km from the river mouth), and increased to 6.0-7.0 downstream from the point 40 km from the river mouth (Figure 2.a). The ratio fluctuated, showing three local maximum values at 32-33 km, 19-23 km and 2-6 km from the river mouth. The ratio at the river mouth, although determined two times, showed the lowest value comparable to the ratio of seawater. Seasonal differences were observed, however, spatial distribution of the ratios revealed an almost uniform distribution irrespective of the investigated season.

The ratio of SO_4^{2-}/Cl^- in the river surface water of the Chikugogawa River exhibited an extremely high value in the uppermost basin, whereas the ratio constantly registered in the range 1.0-2.0 in the area downstream from the point 125 km from the river mouth (Figure 2.b). The values for the estuary of the Chikugogawa River (downstream from the point 20 km from the river mouth) remained lower in comparison to the ratio of the seawater. The lower values for SO_4^{2-}/Cl^- in the estuary revealed seawater inundation at least 20 km from the river mouth, nearly to the floodgate at the 24.5 km point. Although investigation of the Chikugogawa River has been done only three times, the spatial distribution of the SO_4^{2-}/Cl^- ratio revealed rather uniform results irrespective of seasonal difference.

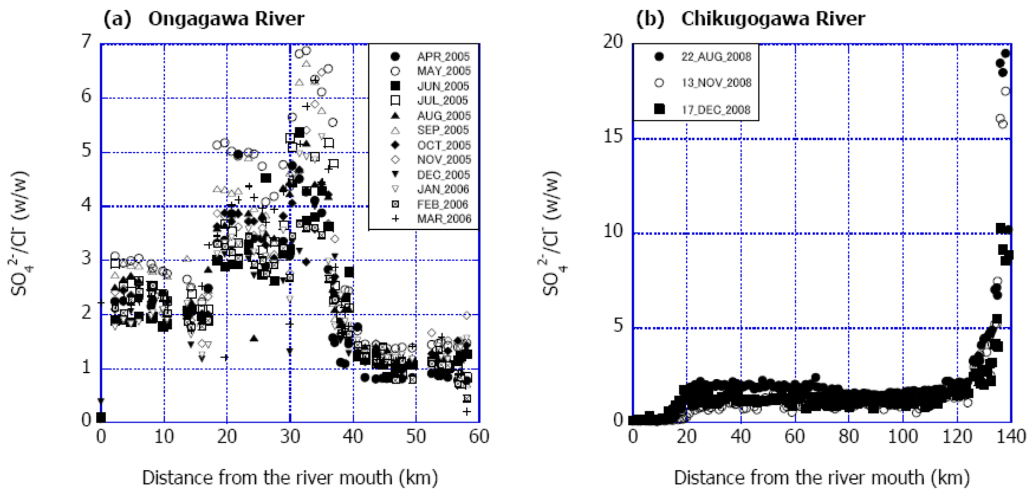

Figure 2. Sulfate ion/chloride ion (w/w) ratios of the river water in northern Kyushu, Japan: (a) the Ongagawa River from April 2005 to March 2006 (12 times) and (b) the Chikugogawa River on 22 August, 13 November, and 17 December 2008 (3 times).

The Ongagawa River basin has many abandoned coal mining sites, and the SO_4^{2-}/Cl^- ratio (w/w) reached as high as 7.0 in the middle basin, whereas the ratios in the Chikugogawa River, which has few coal mining sites in its basin, were constantly in the range 1.0-2.0 in the middle basin, except for the extremely high value (20.0) at the uppermost basin of the river. The upper basin of the Chikugogawa River is in a volcanic region, and consequently the effects of sulfur emission from the active volcanoes as well as the effects of high concentrations of sulfate in spring waters affect the river water, resulting in the higher ratio of SO_4^{2-}/Cl^-. Thus we found that the SO_4^{2-}/Cl^- ratio of the river water adjacent to abandoned coal mining sites was much higher than was the case with a river without coal mining sites in the basin. Pyrite contamination is found in the sediments abandoned after coal mining, and pyrite is thus distributed widely in former coal mining sites. Oxidation of pyrite in such coal mining sites produces sulfuric acid, which is dissolved in the water discharged from the coal mining sites (mine drainage). The SO_4^{2-}/Cl^- ratios for the Ongagawa River water showed rather broad peaks throughout the middle basin of the river, implying the sulfate contamination in the river water is due to non-point sources, probably via the ground water flow system. Compared to the sulfate contamination from the coal mining sites, sulfate discharges from the volcanic area are rather due to a point source. Some part of sulfate would migrate to the river water via ground water flow, because the SO_4^{2-}/Cl^- ratios in the Chikugogawa River registered higher values from the origin to the point 15 km downstream from the origin. However, the SO_4^{2-}/Cl^- ratio at the origin of the Chikugogawa River was extremely high, and thus most of the sulfate migrates to the water directly from the volcanic sites via surface water flow or from spring water. Sulfuric acid produced by pyrite oxidation affects not only the soil itself but also river and lake water systems after discharge from acid sulfate soil (Monterroso and Macías 1998, Blunden et al. 2001). It has been reported that pyrite-containing rocks such as volcanic rocks affect the sulfate ion concentration in stream water discharged from these pyrite-containing rocks (Igarashi et al. 2003). In the tropical peat swamp forest area, peat has decomposed because of a change in land use to agriculture, and the peat layer thickness decreased after this shift. Pyrite included in the sediment under the peat layer is oxidized by the diffused oxygen from the atmosphere via a thin peat layer, and sulfuric acid is thus discharged from the peat land. Water discharged from canals into the main stream of the Sebangau River and the Kahayan River revealed lower pH results, compared to the values found for the mainstream water of the rivers (Haraguchi 2007), implying that sulfuric acid migrates from the canals to the main stream of the rivers as a point source. In the dry season, the SO_4^{2-}/Cl^- ratios in the Sebangau River showed a maximum at the 100 km point (Figure 3.a). The ratios decreased from the 100 km point to the 45 km point and then fluctuated around 0.18, the same value found in the sea water, from the 45 km point to the river mouth. Increases in the SO_4^{2-}/Cl^- ratio values downstream from the 135 km point implied that the effect of pyrite on the river water chemistry appeared downstream from the point 135 km from the river mouth. In the rainy season, the ratios showed the same tendencies as in the dry season, however, the ratios were much higher than during the dry season (Figure 3.b). The SO_4^{2-}/Cl^- ratios started to increase from the point ca. 110 km from the river mouth. The maximum values were found 45 km from the river mouth and 55 km downstream, in comparison to the results uncovered in the dry season. The effect of seawater appeared only in the lower basin in the rainy season because of the high water level of the river, and in addition the SO_4^{2-}/Cl^- ratios were extremely high in the rainy season, compared to those found in the dry season. The SO_4^{2-}/Cl^- ratios in the Kahayan River in the dry season reached a maximum at a point ca. 275 km from the river mouth (Figure 3.c). The ratios decreased from the 275 km point to about the 150 km point. The ratio increased a little downstream at that point, and then the same value as in seawater was found from the 50 km point to the river mouth. Increases in SO_4^{2-}/Cl^- ratios downstream from the 150 km point implied that the effect of pyrite on the river water chemistry appeared downstream from the point 150 km from the river mouth, although the ratios were higher even in the upper stream of the river up to 330 km from the river mouth. In the rainy season, the ratios exhibited extremely high values in the lower basin of the river (Figure 3.d). The SO_4^{2-}/Cl^- ratios started to increase from ca. 110 km from the river mouth, and the maximum appeared at 50 km from the river mouth, much as it did in the Sebangau River. The SO_4^{2-}/Cl^- ratios were slightly higher between the points 240 and 330 km from the river mouth of the Kahayan River in the rainy season, and also relatively higher results were observed in this part of the river during the dry season. Population in this area of the basin is higher than in other areas, and thus these results may occur because of sulfate discharge from the urbanized area. Intensive land use changes from peat swamp forest to urbanization in the area could also be accelerating pyrite oxidation in the basin.

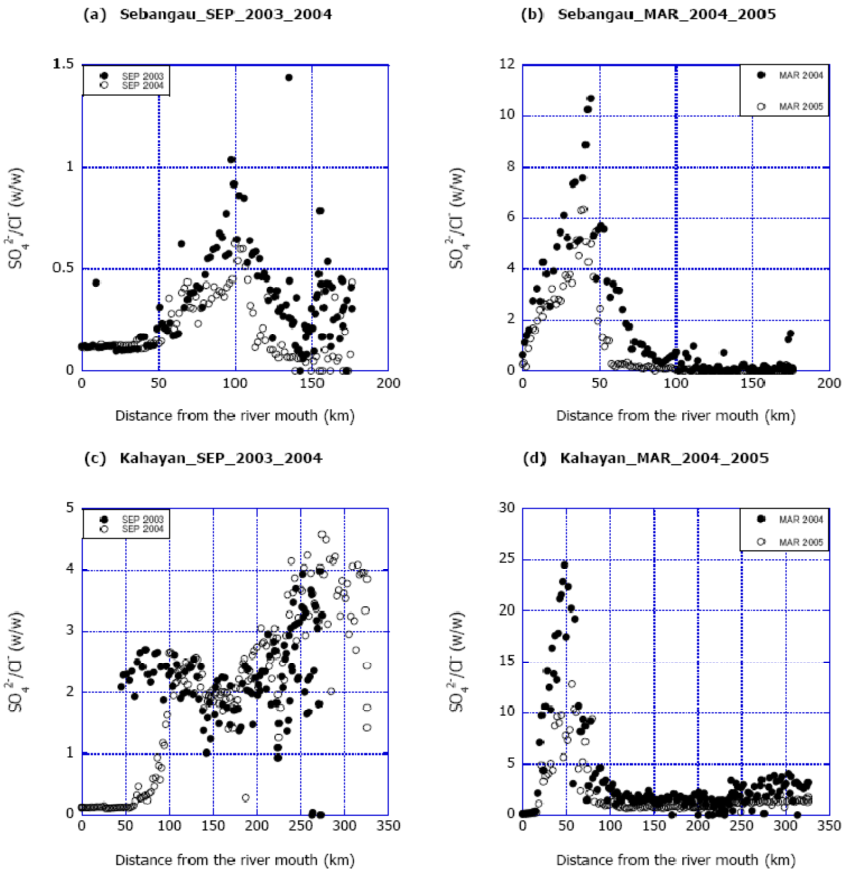

Figure 3. Sulfate ion/chloride ion (w/w) ratios of the river water in Central Kalimantan, Indonesia: (a) the Sebangau River in the dry season in September 2004 and 2005 (2 times), (b) the Sebangau River in the rainy season in March 2004 and 2005 (2 times), (c) the Kahayan River in the dry season in September 2004 and 2005 (2 times) and (d) the Kahayan River in the rainy season in March 2004 and 2005 (2 times).

The SO_4^{2-}/Cl^- ratio can be used as a parameter for estimating the contribution of pyritic sulfate to river water chemistry, and this parameter demonstrated that sulfuric acid loading from pyrite oxidation had appeared from the river mouth up to a point 135-150 km upstream in both the rivers in the tropical peat swamp forest. Water of the mainstream of the rivers as well as water discharged from artificial canals into the mainstream in the rainy season showed much higher acidity and a higher SO_4^{2-}/Cl^- ratio than they did in the dry season. This implies that the discharge of pyritic sulfate from peat swamp forests into the freshwater system is much higher in the rainy (high water table) season than the dry (low water table) season. Water in the canals in the rainy season was found to be highly acidic (pH = 2.0-3.0).

Sulfuric acid pollution in open mining casting lakes after brown coal mining is a serious environmental problem in some areas of Europe. Surface mining of brown coal may have severe impacts on the quality of surface and underground waters due to the possible formation of acid mine drainage. Pyrite oxidation and secondary reactions may result in a solution pH < 2 and large concentrations of SO_4^{2-}, Fe and Al in the leaching water. At sites with low proton buffer capacity, there is a concern that the toxicity of Al and heavy metals may restrict reforestation efforts (Schippers et al. 2000). When carbonate is distributed in the basin, H^+ formed by pyrite oxidation is rapidly buffered by $CaCO_3$. However, pyrite oxidation accompanied by carbonate weathering forms gypsum and this affects the hydrological properties of the aquifer (Ritsema and Groenenberg 1993).

Contamination by sulfate of a river system is observed in many ecosystems, including peatlands, mangroves and mining sites. Parts of the discharged water contaminated with sulfate flow into river system via ditches as point sources and in such cases the discharging process is easy to determine. However, sulfate migration via ground water system is complicated and it is not easy to evaluate its effects on river systems. The ratio of SO_4^{2-}/Cl^- yielded higher values in the basin of the tropical peat swamp forest with acid sulfate soils, as well as in the abandoned coal mining sites. Consequently, the ratio is commonly used as an index for the qualitative evaluation of non-point contamination of sulfate in river systems. Quantitative analyses of sulfate contamination of river systems, as well as impacts of sulfate contamination on the freshwater community, are the next steps in our research.

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Chapter 9

RELATION BETWEEN RIVER MANAGEMENT AND ECONOMIC GROWTH IN URBAN REGIONS

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ABSTRACT

Water stress in northern China has intensified water use conflicts between upstream and downstream areas and also between agriculture and municipal/industrial sectors year by year. In this study, the NIES Integrated Catchment-based Eco-hydrology (NICE) model (Nakayama, 2008a, 2008b, 2010a, 2010b; Nakayama and Fujita, 2010; Nakayama and Watanabe, 2004, 2006, 2008a, 2008b; Nakayama *et al*., 2006, 2007, 2010) was applied to the Biliu River catchment, northern China, to estimate the carrying capacity of the water resource there. The model reasonably backcasted the degradation of water resources such as river discharge and groundwater after the completion of the reservoir in the middle reach of the river. The normalized difference vegetation index (NDVI) calculated from NOAA/AVHRR satellite image clearly showed vegetation degradation downstream of the reservoir. Furthermore, statistical analysis of a decoupling indicator (OECD, 2001) based on the water carrying capacity simulated by NICE and on the satellite data of vegetation index indicated that water-related stress in Dalian city, where the economy has grown rapidly after the completement of the reservoir, has increased in accordance with the environmental degradation below the reservoir. These results indicate a close relationship between water resource and economic growth, which has greatly affected ecosystem degradation and its serious burden on the environment in the catchment. The simulated results also highlight the linkage between urban development in Dalian and sustainable water resource management, and this assessment of the interactions between the sites of water source and demand would support decision-making on sustainable development in the catchment.

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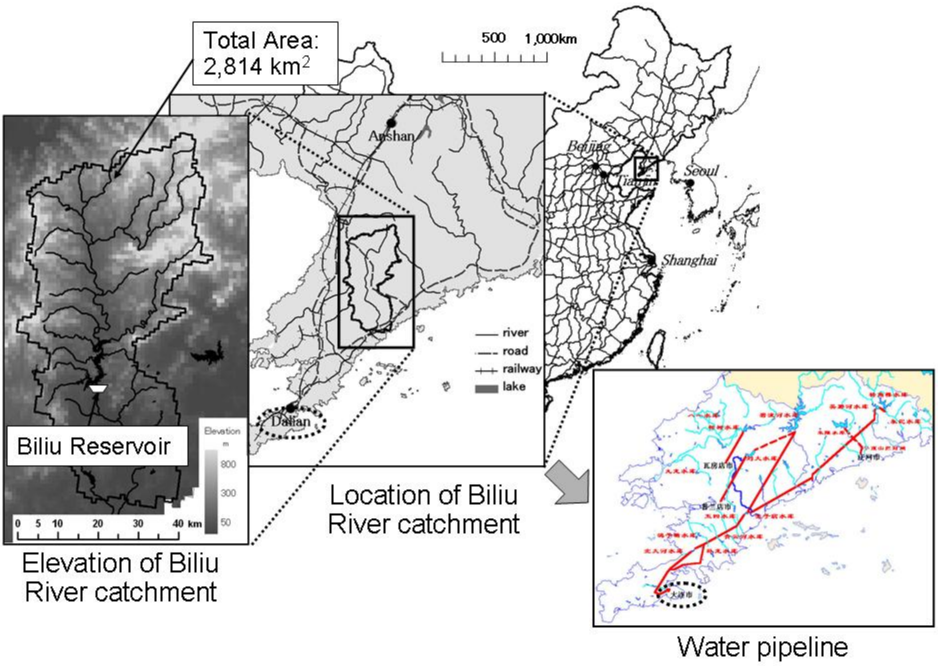
1. INTRODUCTION

Water stress in northern China has intensified water use conflicts between upstream and downstream areas and between agriculture and the municipal/industrial sectors (Nakayama et al., 2006; Nakayama, 2010b). Some researchers stated that China's environmental pressure already exceeds its carrying capacity of this densely populated land (Niu and Harris, 1996; Varis and Vakkilainen, 2001).

Over 20 000 smaller catchments in China play an important role in supporting the local development and maintaining local ecosystem functions (Geng *et al*., 2010), but few studies have looked at changes in water and heat dynamics, mainly on account of the lack of data for quantifying complex phenomena related to human activities.

The Biliu River catchment (2814 km^2) is located in the southern part of Liaoning province, a heavy industrial base in northeast China (Figure 1). The Biliu has a total length of 156 km, and is the primary water source for Dalian city, which has a population of over six million people and thousands of industries. Dalian has a maritime climate and inherently little fresh water supply. Its fast economic growth has been supported by increased consumption of water from further field (Figure 2). Though Dalian entered a high-growth period and GDP increased more than tenfold during 1992–2007 thanks to overseas funding (Dalian Planning Committee, 2001), water consumption increased much more slowly and the conflict between water supply and demand has tended to worsen in recent years (Zhang and Xiong, 2006). The shortage of urban water greatly influences the development of the city and the lives of its residents, and has become the main obstruction to sustainable development of economy and society (Han *et al*., 2008).



Figures 1. Location of study area and elevation in the Biliu River catchment and Dalian city. Bold black line is the border of catchment.

Since the completion of the Biliu Reservoir in 1984, most of the water has been piped to Dalian, and the catchment has changed from water-rich to water-poor (Figure 2c), as evidenced by drying out of the downstream river, groundwater fall, and seawater intrusion, all seen in other regions of northern China also (Qian and Zhu, 2001; Ren *et al.*, 2002; Nakayama *et al.*, 2006; Nakayama, 2010b). In recent years, Dalian government made a local circular economy implementation plan in Dalian city in 2006, which aimed to cut water consumption about 20% per million GDP (RMB) in 11th five-year plan for Dalian city's economic and social development (Dalian City Government, 2006). How to meet the water demands of its swelling city and industry without undermining both its own agriculture and the local ecosystem is the biggest challenge for the Dalian government.

The objective of this research was to evaluate the linkage between urban development in Dalian and sustainable water resource management in the Biliu River catchment. First, we applied the process-based NIES Integrated Catchment-based Eco-hydrology (NICE) model to the Biliu River catchment to estimate the changes in water resources since the completion of the Biliu Reservoir. Secondly, we used NOAA/AVHRR satellite image analysis to evaluate the simulated result relative to the environmental degradation in the lower region of source area. Thirdly, we statistically analyzed a decoupling indicator based on the estimated water carrying capacity to evaluate water-related stress against economic development in Dalian. This assessment of the interactions between the sites of water source and demand will support decision-making on sustainable development in the catchment.

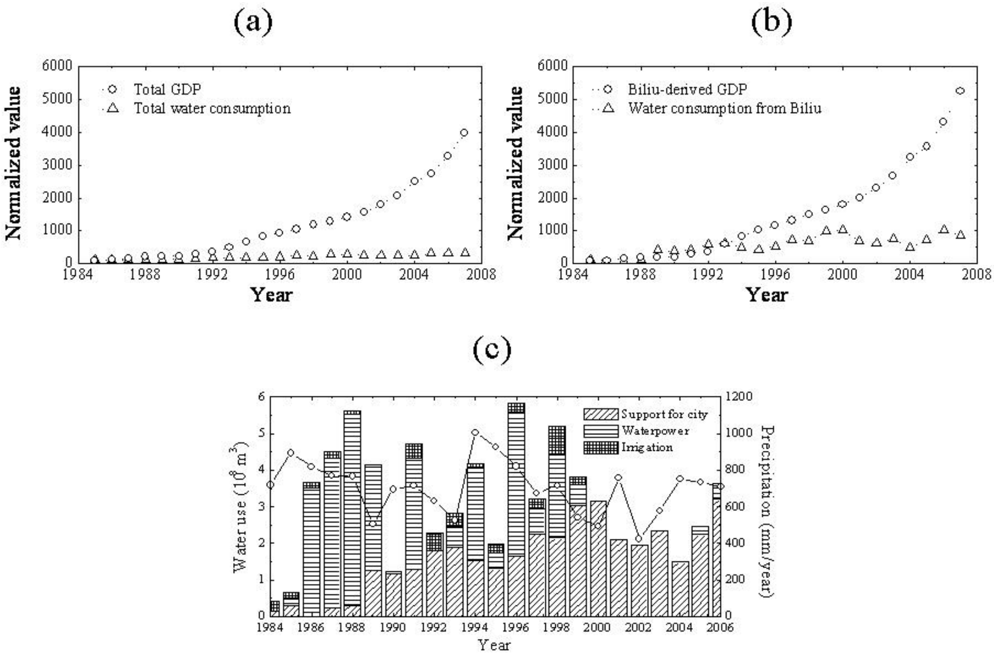


Figure 2. Annual time-series of GDP and water consumption at Dalian city after the completement of Biliu Reservoir (Dalian Water Resource Bureau, 2007a); (a) GDP and water consumption in the entire Dalian city, (b) GDP and water consumption in the urban area of the city, and (c) water intake from Biliu Reservoir, respectively. GDP and water consumption were plotted as the values based on those in 1985 (value at 1985 = 100) in Figures 2 (a)-(b). Rainfall anomalies at the reservoir are also plotted in the right y-axis in Figure 2 (c).

2. METHOD

2.1. NICE Model

Our group has developed the NICE (NIES Integrated Catchment-based Eco-hydrology model) series of process-base catchment models applicable to natural, agricultural, and urban regions in the catchment (Nakayama, 2008a, 2008b, 2010a, 2010b; Nakayama and Fujita, 2010; Nakayama and Watanabe, 2004, 2006, 2008a, 2008b; Nakayama *et al*., 2006, 2007, 2010). The NICE models connect surface-unsaturated–saturated water processes and land-surface processes describing variations in phenology based on MODIS (Moderate Resolution Imaging Spectroradiometer) satellite data (Figure 3). The submodels consider water and heat fluxes from the ground to the surface; for example, the gradient of hydraulic potentials between the deepest layer of unsaturated flow and the groundwater level; the effective precipitation calculated from actual precipitation, infiltration into the upper soil moisture store, and evapotranspiration; and the seepage between river water and groundwater. Details are available in Nakayama and Watanabe (2004).

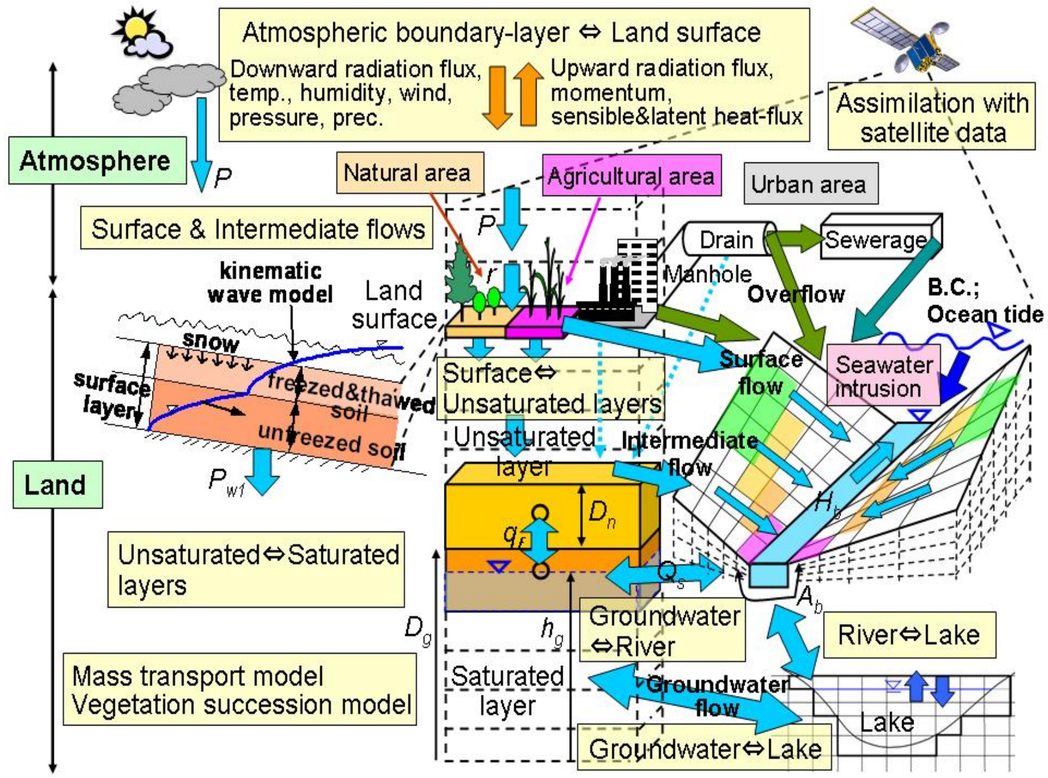


Figure 3. NIES Integrated Catchment-based Eco-hydrology (NICE) model.

2.2. Satellite Analysis

The NDVI was calculated from NOAA/AVHRR satellite data with a resolution of 1.1 km collected over East Asia during 1984–2007. The original data were downloaded from WebPaNDA (2008) and re-sampled to change the resolution by the nearest neighbor method. The data area was selected to match the simulation area (39.35°–40.50°N, 122.05°–123.00°E). To minimize the effects of clouds and atmospheric contaminants, we used composite data within a 10-day or monthly period that had the fewest clouds, as identified by the highest NDVI value. The NDVI was calculated as:

$$NDVI = (NIR - RED)/(NIR + RED) \tag{1}$$

where RED is band 1 (visible, 0.58–0.68 μm), and NIR is band 2 (near-infrared, 0.725–1.10 μm). NDVI can be used as an indicator of vegetation stress, particularly that due to water shortage (Singh *et al.*, 2003). The important point of this paper is that economic growth in the urban area (through water consumption) has a close relationship with environmental degradation in the water source area (downstream of Biliu River catchment) shown in the NDVI trend in the following.

2.3. Statistical Analysis of Decoupling Indicator

To identify impacts of social activities (=driving force) and their pressures such as water consumption and economic growth on the environment, we used a decoupling indicator procedure to analyze them in Dalian city. The OECD has developed procedures to decouple environmental pressures from economic growth (OECD, 2001). A decoupling indicator is calculated from the ratio of Pressure to Driving Force at the end to the value at the start of a given time period:

Decoupling indicator = 1 – (EP/DF)end of period / (EP/DF)start of period (2)

where *EP* is environmental pressure and DF is driving force. The decoupling indicator is zero or negative in the absence of decoupling, positive in the relative decoupling, and has a maximum value of 1 when environmental pressure reaches zero (absolute decoupling). *EP* is the annual Biliu-derived water consumption in all of Dalian or in the urban area of Dalian in a time series from 1985 to 2007, and the water consumption of heavy industries or light industries derived from the Biliu River from 2001 to 2007. *DF* is the GDP in Dalian or in the urban area. In our analysis of the impact of water abstraction from the Biliu River on the environment in the catchment, *EP* is the NDVI as an index of environmental degradation and *DF* is water supply from the Biliu River.

3. Input Data and Boundary Conditions for Simulation

3.1. Input Data

Six-hour reanalysis data with a resolution of 1° × 1° were assimilated with daily observation data of solar radiation, precipitation, temperature, humidity, and wind speed at a weather station operated by Dalian Water Resource Bureau (2005–2006) and Yingkou City Government (2005–2006) (Table 1) to create grid data with a resolution of 1 km, and to input into each grid cell for a 2005–2006 simulation by interpolating these parameters in inverse proportion to the distance back-calculated in each cell. The mean elevation of each 1-km grid cell was calculated by using the spatial average of a global digital elevation model (DEM; GTOPO30) with a horizontal grid spacing of 30 arc-seconds (approximately 1-km mesh) (US Geological Survey, 1996) throughout the Biliu River catchment (Figure 1). Vegetation class and soil texture were categorized and digitized into 1-km mesh data from the Vegetation and Soil Maps of China (1:4 000 000) (Chinese Academy of Sciences, 1988) and the Soil Map of Dalian (1:100 000) (Liaoning Geology Investigation Agency, 2007). The geological structures were decided from scanned and digitized geological material (Geological Atlas of China, 2002) to divide geological structures into four types as described by Nakayama *et al.* (2010). Digital land cover data produced by the Chinese Academy of Sciences (CAS) (1988) based on Landsat TM data from the early 1990s were categorized and divided into 1-km mesh data for the simulation. Details are given in Nakayama and Watanabe (2008b). Forest, grass, and scrub cover the mountainous and hilly areas in the upper catchment. Cultivated fields are widely distributed in the valley and lower regions. The water consumption in Dalian during 1985–2007 was compiled from statistical data (Dalian Water Resource Bureau, 2000–2007). Annual GDP data during 1985–2007 came from the Dalian Planning Committee (2001), the Dalian City Government (1990–2007), and the Dalian Bureau of Statistics (2002–2007). Details are available in Nakayama *et al.* (2010).

3.2. Boundary Conditions and Running Simulation

At the upstream boundaries, the reflecting condition on the hydraulic head was used in the groundwater flow submodel on the supposition that there is no inflow from the mountains in the opposite direction (Nakayama and Watanabe, 2004). At the southern sea boundary (Yellow Sea), a variable head was set by using observed monthly data (T-1 in Table 1; Dalian Water Resource Bureau, 2005–2006). The simulation area is 60 km wide by 110 km long on the basis of the Albers (WGS 1984) co-ordinates, covering almost all of the Biliu River catchment (Figure 1). The area was divided into a grid of 60 × 110 blocks with a grid spacing of 1 km in the horizontal directions and into 20 layers with a weighting factor of 1.1 (finer at the upper layers) in the vertical direction. The upper layer was set at 2 m depth, and the 20th layer was defined as an elevation of 200 m below sea level. Simulations covering 1 January 2005 to 31 December 2006 were run for calibration. A time step of Δt = 1 h was used. Simulations were validated against observed river discharge at 5 points, groundwater level at 3 points, and soil temperature and moisture at 1 point (Dalian Water Resource Bureau, 2005–2006; Yingkou City Government, 2005–2006) (Table 1).

Table 1. Lists of observation stations for validation

No.	Point Name	Type	Lat.	Lon.	Elev.(m)
R-1	Xiao Shipeng	Rainfall Gage	40°17.1'	122°27.0'	199.0
R-2	Meng Jiadian	Rainfall Gage	40°14.6'	122°31.5'	191.0
R-3	Kuang Donggou	Rainfall Gage	40°10.3'	122°43.0'	233.0
R-4	Taiping Zhuang	Rainfall Gage	40°8.7'	122°44.0'	227.0
R-5	Xi Pashan	Rainfall Gage	40°7.3'	122°32.3'	115.0
R-6	Biliu Reservoir	Rainfall Gage	39°49.2'	122°29.5'	53.0
No.	Point Name	Type	Lat.	Lon.	Elev.(m)
R-7	Guiyunhua Village	Rainfall Gage	39°55.7'	122°35.9'	80.0
R-8	Daiiang Village	Rainfall Gage	40°1.7'	122°42.6'	219.0
R-9	Shifogou Village	Rainfall Gage	39°47.6'	122°38.4'	84.0
R-10	Tianyi	Rainfall Gage	39°54.6'	122°26.4'	133.0
R-11	Shuangta Village	Rainfall Gage	39°46.2'	122°28.7'	27.0
R-12	Xiaosongjia Village	Rainfall Gage	39°34.3'	122°32.8'	9.0
R-13	Jian Chang	Rainfall Gage	40°1.7'	122°32.3'	91.0
W-1	Gaizhou	Weather Data	40°23.7'	122°22.1'	8.0
W-2	Biliu Reservoir	Weather Data	39°49.2'	122°29.5'	53.0
W-3	Zhuanghe	Weather Data	39°42.0'	122°59.5'	3.0
W-4	Pulandian	Weather Data	39°23.7'	121°58.0'	2.0
W-5	Dalian	Weather Data	38°54.7'	121°36.1'	14.0
G-1	Jian Chang	Groundwater Level	40°1.7'	122°32.3'	91.0
G-2	Biliu Reservoir	Groundwater Level	39°49.2'	122°29.5'	53.0
G-3	Xiaosongjia Village	Groundwater Level	39°34.0'	122°33.0'	9.0
D-1	Jian Chang	River Discharge	40°1.7'	122°32.3'	91.0
D-2	Lingxi	River Discharge	39°55.7'	122°35.9'	80.0
D-3	Biliu Reservoir	River Discharge	39°49.2'	122°29.5'	53.0
D-4	Downstream of Biliu Reservoir	River Discharge (analyzed)	39°48.8'	122°29.6'	33.0
D-5	Xiaosongjia Village	River Discharge	39°34.3'	122°32.8'	9.0
S-1	Jian Chang	Soil Temperature and Moisture	40°1.7'	122°32.3'	91.0
T-1	Chengzitan	Tidal Level	39°29.2'	122°33.9'	5.0

4. RESULTS

4.1. Hydrologic Cycles and Vegetation Change Affected by the Construction of Biliu Reservoir

The simulated results of river discharge and groundwater level were firstly calibrated for the Biliu River catchment during 2005–2006. NICE backcasted the river discharge downstream of the reservoir after its completion from the total volume of water piped to Dalian (Figure 4.a).

The simulated result clarified that the river discharge has decreased year by year since the completion of the reservoir as water demand has increased in the urban area. It can be clearly seen that most of the water in the river is already being used at the beginning of this century.

Because the rapid development of industry and urbanization and the increase in farmland irrigation have also increased water demand to the limit of the carrying capacity (Varis and Vakkilainen, 2001), the groundwater table has also declined in the floodplains of the river's downstream areas.

To study spatial patterns in NDVI trends in the catchment during 1984–2007, we estimated the linear trend of annual mean NDVI over the study period using ordinary least-squares regression (Figure 4.b).

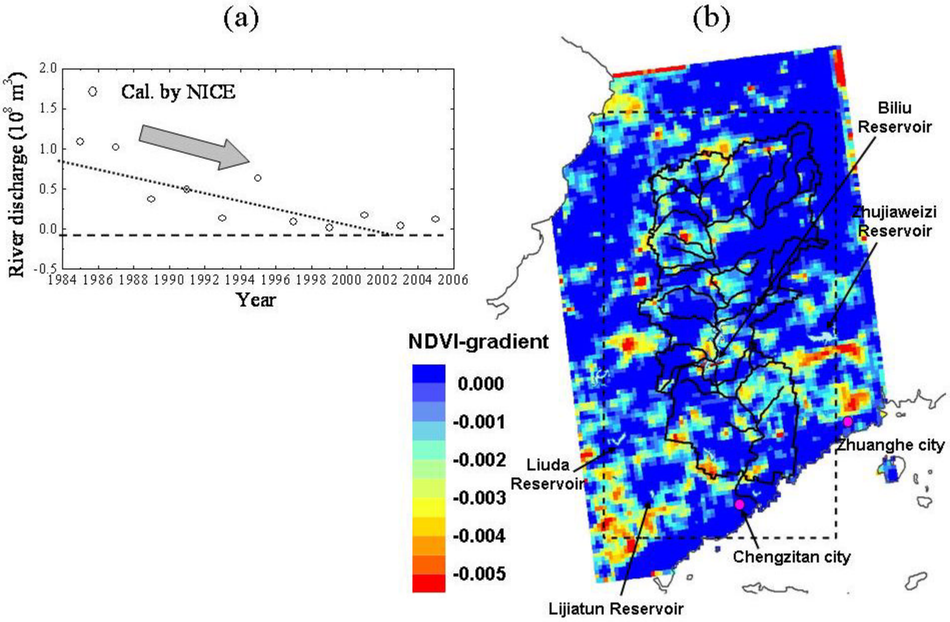


Figure 4. Hydrologic cycles and vegetation change affected by the reservoir; (a) simulated river discharge at downstream of the reservoir, and (b) spatial patterns in NDVI gradient in the entire catchment during 1984-2007. Dashed-square is the border of simulation area as shown in Figure 1.

A significant decrease in the NDVI occurred primarily downstream of the reservoir, as well as downstream of the Zhujiaweizi and Liuda reservoirs, which were built in 1968 and 1971 respectively (Dalian City Government, 2003). Downstream of the Zhujiaweizi Reservoir and nearby, the NDVI decreased as a result of several effects, including a reduction in water resources and rapid urbanization surrounding Zhuanghe city (Wang, 1997). Mountainous areas upstream of the Biliu Reservoir have higher NDVI values than downstream areas. This difference is closely related to the water shortage and seawater intrusion since the construction of the Biliu Reservoir, which has greatly reduced crop growth and production in coastal areas.

4.2. Relationship between water resources and urban activity

The economy has grown faster than water consumption (Figure 2), which shows a pronounced relative decoupling between economic growth and environmental pressure in Dalian during 1992–2007. The consumption of water derived from the Biliu River shows a relative decoupling from the GDP of the urban area (Figure 5.a). Decoupling indicator value of the urban area was higher than that of the entire Dalian city in 1992–1993, 2003–2004, and 2006–2007, when the environmental pressure from local area was much less than that of Dalian city, although the values in 1985–1988 were irregular owing to the start of dam operation. The decoupling indicator values declined, were unstable, and were negative twice in 8 years (1998–1999 and 2004–2005), although the Dalian government has increased the water price several times to control water consumption (Dalian Water Resource Bureau, 2000–2007).

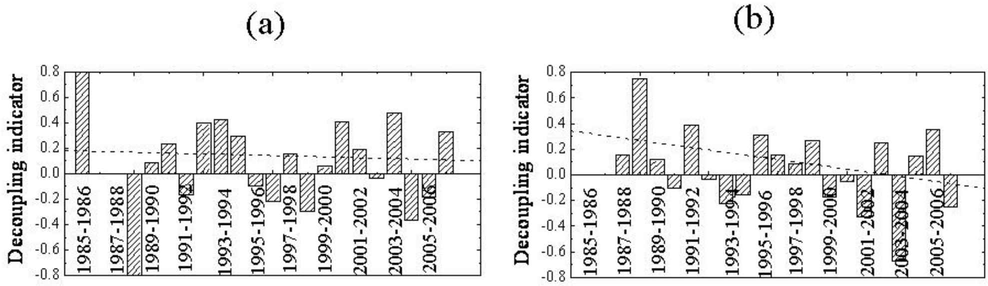


Figure 5. Annual trends of decoupling indicators between; (a) water consumption and GDP in the urban area, and (b) NDVI and water consumption, respectively. Dashed-lines are least-square regression lines estimated from annual decoupling indicators (bars).

The value dropped from 0.43 in 1993–1994 to –0.37 in 1998–1999, became positive at 0.33 again in 2003–2004, and then declined again. These trends show that the environmental pressure of water consumption has increased with economic growth in the urban area, although relative decoupling predominated, especially when the decoupling indicator value was negative. The impact of water withdrawal from the Biliu Reservoir on environmental degradation of NDVI (Figure 4.b) downstream is shown in Figure 5.b. The decoupling indicator value gradually decreased from 0.75 (relative decoupling) in 1988–1989 to –0.25 (no decoupling) in 2006–2007 although the periods of 1985-1988 take irregular values due to the start of dam operation and incorrect data. This shows increasing environmental degradation with increasing abstraction of water from the Biliu River. These results indicate that the environmental degradation in the Biliu River catchment will grow more serious with economic growth in the coming years.

5. Discussions and Conclusions

NICE correctly backcasted the gradual decrease in water resources since the completion of the reservoir in the middle of the catchment and its relation to the reduced NDVI downstream of the reservoir. The environmental degradation downstream of the reservoir is closely related to the decrease in river discharge and groundwater level (Figure 4), all characteristic of catchments in northern China (Qian and Zhu, 2001; Ren *et al*., 2002; Nakayama *et al*., 2006; Nakayama, 2010b). This degradation has been caused by the high economic growth in Dalian and the resultant degradation of production-support and life-support systems as indicated by the statistical analysis of decoupling indicators (Figure 5.a), in the same ways as the previous research (Niu and Harris, 1996). This result implies that the high increase of water consumption has not necessarily offset the high increase of GDP in the urban area, which is the situation that water is yet insufficient in spite of the high economic growth.

Unfortunately, the technique of saving water has not improved so much in this study area because the cost of potable water is still cheap and therefore the economy depends greatly on the water-derived consumption. In the same reason, the local companies do not want to afford a higher price of desalination in addition to the difficulty to treat the solid wastes generated from desalination process as well as the very limited landfill space. Decoupling analysis between vegetation index and water consumption also showed water consumption dependent industrial structure in Dalian city has caused deterioration in the downstream area of Biliu River catchment with higher pace than the increasing water consumption (Figure 5.b). These results indicate that the environmental pressure has not been considered in Dalian by either municipal governments or enterprises as a matter of crisis for the carrying capacity of water resources unlike more serious situation in other northern planes regions of China (Niu and Harris, 1996; Varis and Vakkilainen, 2001), and that it is urgently necessary to find a better strategy to achieve sustainable development.

As circular economy law by Chinese government and consequent policy by the municipal government of Dalian City and Liaoning Province, orientation for more ecologically efficient industries may seek for the industrial restructuring for sustainably eco-friendly industrial structures (Dalian City Government, 2006). The multi-source water reasonable allocation in the city would be an available solution to improve water use efficiency and achieve sustainable development of urban water resources while minimizing adverse impacts on the environment. The use of seawater, the reuse of treated wastewater, and the collection and use of rainwater from roofs and urban pavements have been discussed by the local government and are gradually increasing in Dalian (Han *et al*., 2008). It is further useful to predict the allocation of multisource water under the benefit of socio-environment optimization, such as, various scenarios like economic benefit maximization, green area maximization, and sewage drainage minimization, et al. The interdisciplinary research compiling scientific assessment and economic activities would provide shaper priority setting for the municipal governments and enterprises on sustainable development in the catchment.

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