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River Ecology

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Abstract Rivers have many service functions such as water supply, food production, sightseeing, and shipping, hence playing an important role in people's living and agricultural production. During the last decades, intensive human activities have been threatening river ecosystem. A better understanding of ecological stresses and assessments of river ecosystem is of great significance to river conservation and management. This chapter discusses river ecology, disturbances to the ecology, and assessments of river ecosystem.

Key Words River ecology • Ecological stresses • Ecological assessments • Biodiversity • Habitat.

1. RIVER ECOSYSTEMS

1.1. Background Information of Rivers

Rivers are such an integral part of the land, and they are much more than merely water flowing to the sea. Rivers carry downhill not just water, but just as importantly sediments, dissolved minerals, and the nutrient-rich detritus of plants and animals. The main functions of rivers are draining floods, supplying drinking water, maintaining ecology, irrigating farmland, transporting sediment, supplying power, providing habitat for fishes, assimilating wastewater, and providing navigation. Humans exploit the resources of rivers by constructing dams and water-diverting channels, developing navigation channels, and harvesting fishes, which result

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in changes in the river hydrology, runoff, sediment transport, riparian and stream habitats, and water quality.

Rivers can be recognized as mountain rivers, alluvial rivers, and estuaries. A mountain river is the most upstream part of the river, including the river source and the upstream tributaries of the river, where the river system flows through mountainous areas and the flow is confined by mountains. Erosion control and vegetation development over the watershed, landslides and debris flows, and control of channel bed incision are major topics of mountain river studies. An alluvial river is defined as a river with its boundary composed of the sediment previously deposited in the valley, or a river with erodible boundaries flowing in self-formed channels. Sediment transportation, water resources development, and flood defense are the most important issues in the alluvial river management. The estuary is the connection part of a river with the water body (lake, sea, or ocean) into which it flows, including the river mouth, a river section affected by the tide, and the water body area affected by the river flow. Delta and coastal processes, eutrophication, and algal blooms are the major challenges for the management of estuaries.

River ecology is the science of studying the relations among different organisms and the relations between organisms and their environment in rivers. In recent years, river ecosystems have been facing the threat of eutrophication and the destruction of natural hydrologic regime. The former is due to the discharge of sewage, while the latter mainly due to the construction of hydraulic engineering. At present, in the world, over 40,000 large dams (>15 m high) impound the rivers, and about 300 new large dams are currently built every year, particularly in the developing countries [1, 2]. The construction of hydraulic engineering can affect water temperature, water chemistry, sediment transport, and vegetation assemblages, thereby threatening the ecological functions of the rivers. Therefore, it is vital to assess ecological status of river systems and to put forward river strategies of conservation and restoration.

1.2. 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 Fig. 3.1. 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:

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 it is forest or agriculture.

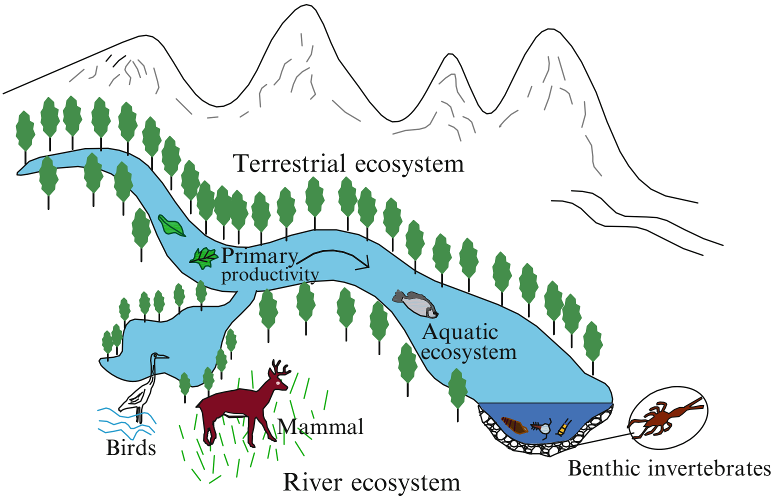


Fig. 3.1. A river ecosystem consists of the terrestrial ecosystem and the aquatic ecosystem, which is affected by and impacts on the landscape ecosystem through input and output (after FISRWG 1997).

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 3.2 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

Fig. 3.2. (**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).

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 [3]. 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 3.3 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

Fig. 3.3. (**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.

the channel, and passive recreation areas such as picnic grounds. Figure 3.4 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: (a) vegetative species clearly different from nearby areas and (b) species similar to adjacent areas but exhibiting more vigorous or robust growth forms. Riparian areas are usually transitional between wetland and upland.

1.3. Ecological Conditions

1.3.1. 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 [4–6]. Flow velocity affects the deliverance of food and nutrients to organisms; however, it can also dislodge them

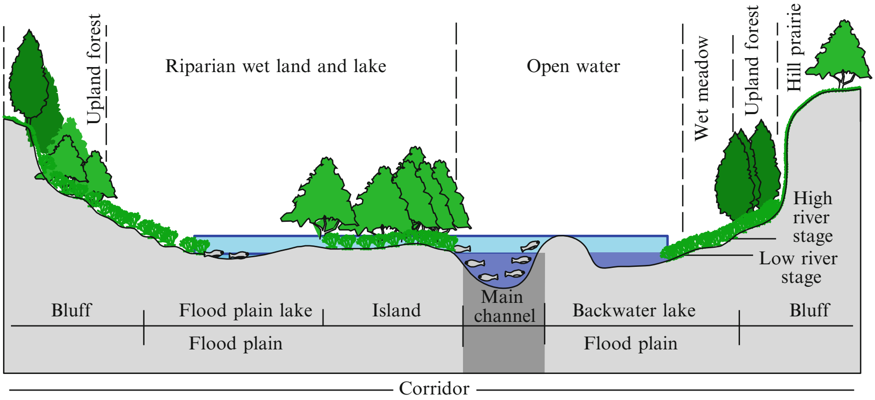


Fig. 3.4. A cross section of a river corridor, in which the river corridor is subdivided by structural features and plant communities (after FIRSWG 1997).

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 [7]. High flows are cues for timing migration and spawning of some fish. When fish detect high flows, some will migrate and some will spawn.

1.3.2. *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 [6, 8]. This can also be applied to algae and fish for their growth rates increase with increased water temperature [5, 9]. Some species are only found in certain areas due to the correlation between temperature and growth, development, and behavior.

1.3.3. *Riparian Vegetation*

Decreased light and temperature in steams can be a result of riparian vegetation [10]. 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 °C when flowing through an open area in direct sunlight but then decreased significantly during flow through 500 m of forest [11]. 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 2–6 °C usually altered key life-history characteristics of some species [12]. 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.

1.3.4. Oxygen

Oxygen enters the water by absorption directly from the atmosphere and by plant photosynthesis [13]. 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 [13]. Dissolved oxygen concentrations of 3 mg/L or less have been shown to interfere with fish populations for a number of reasons [13]. When the oxygen needed for chemical and biological processes exceeds the oxygen provided by reaeration 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 [14].

Pollution that depletes the stream of oxygen has a marked effect on stream communities [15]. 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 [14]. A level of 5 mg/L or higher of dissolved oxygen in water is the level associated with normal activity of most fish [16]. In streams filled with trout, the dissolved oxygen concentration has been shown, by analysis, to be between 4.5 and 9.5 mg/L [14].

1.3.5. pH Value

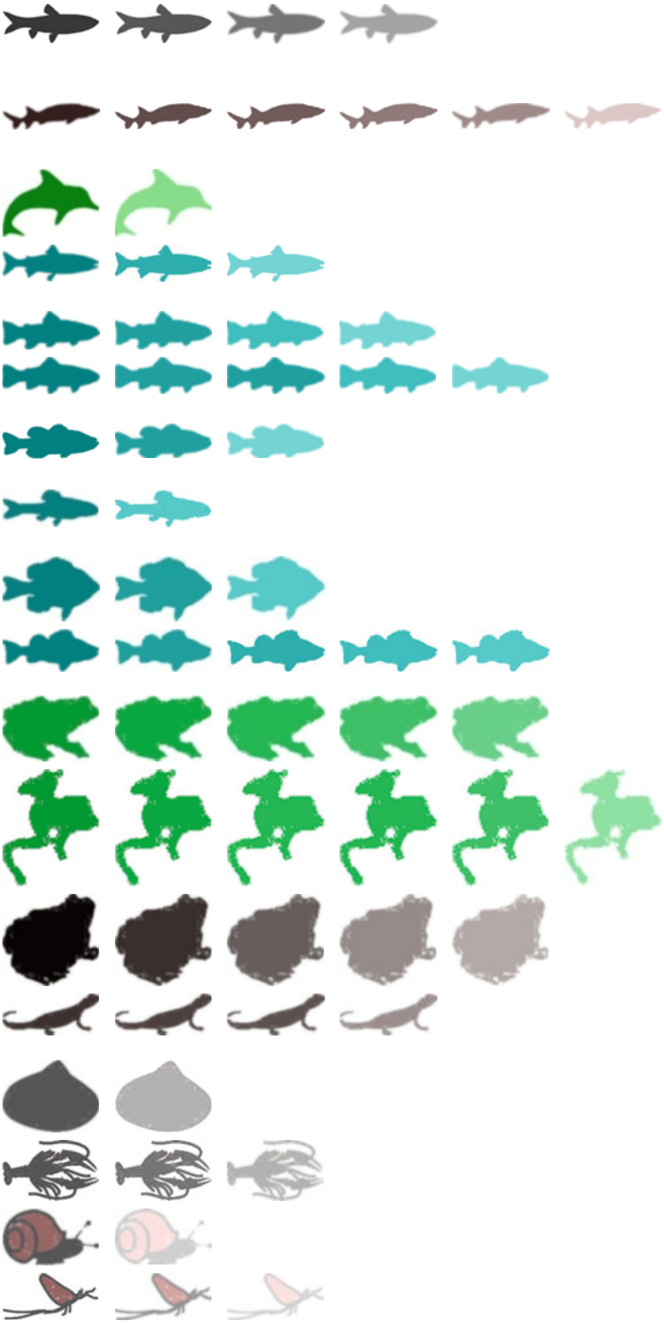
Aquatic biota survive best when the water has a pH of 7, i.e., nearly neutral hydrogen ion activity. If the pH changes, either becoming more acidic or more alkaline, the stress levels increase and eventually species diversity and abundance decrease. In streams under the stresses of various human activities, the pH often becomes more acidic and many species suffer, as shown in Table 3.1 (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 [17]. Some soils have the ability to buffer pH changes; however, those which cannot neutralize acid inputs cause environmental concerns.

1.3.6. 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 [18–20]. 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 Fig. 3.5.

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

Table 3.1
Effects of acid rain on some aquatic species



pH	6.5	6.0	5.5	5.0	4.5	4.0	3.5	3.0
Grass carp (*Ctenopharyngodon idellus*)	●	●	●	●				
Chinese sturgeon (*Acipenser sinensis*)	●	●	●	●	●	●		
Chinese river dolphin (*Lipotes vexillifer*)	●	●						
Rainbow trout (*Oncorhynchus mykiss*)	●	●	●					
Brown trout (*Salmo trutta*)	●	●	●	●				
Brook trout (*Salvelinus fontinalis*)	●	●	●	●	●			
Smallmouth bass (*Micropterus dolomieu*)	●	●	●					
Fathead minnow (*Pimephales promelas*)	●	●						
Pumpkinseed sunfish (*Lepomis gibbosus*)	●	●	●					
Yellow perch (*Perca flavescens*)	●	●	●	●	●			
Bullfrog (*Rana catesbeiana*)	●	●	●	●	●			
Wood frog (*R. sylvatica*)	●	●	●	●	●	●		
American toad (*Bufo americanus*)	●	●	●	●	●			
Spotted salamander (*Ambystoma maculatum*)	●	●	●	●				
Clam	●	●						
Crayfish	●	●	●					
Snail	●	●						
Mayfly	●	●	●					
pH	6.5	6.0	5.5	5.0	4.5	4.0	3.5	3.0

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).

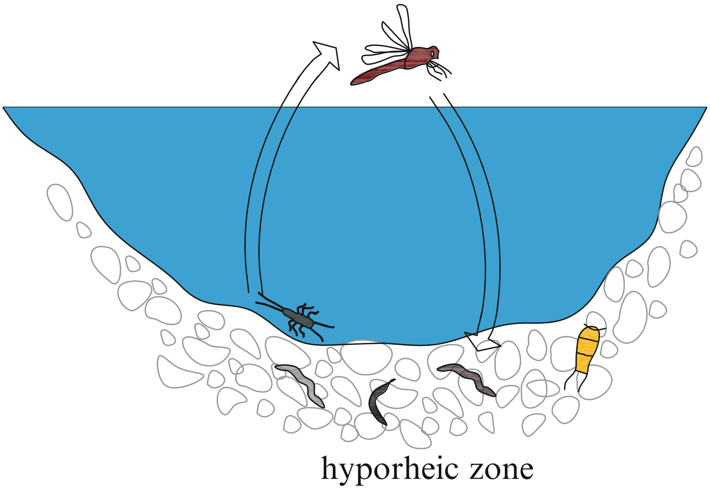


Fig. 3.5. Schematic diagram of hyporheic zone.

provide for production, decomposition, and other processes [21]. 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 [15]. If woody debris, from nearby trees in forests and riparian areas, fall into the stream, the quantity and diversity of aquatic habitats are increased [22, 23].

1.3.7. 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 streambed, 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.

1.4. Biological Assemblages

Stream biota are often classified into seven groups—bacteria, algae, macrophytes (higher plants), protists (amoebas, flagellates, ciliates), microinvertebrates (invertebrates less than 0.5 mm in length, such as rotifers, copepods, ostracods, and nematodes), macroinvertebrates

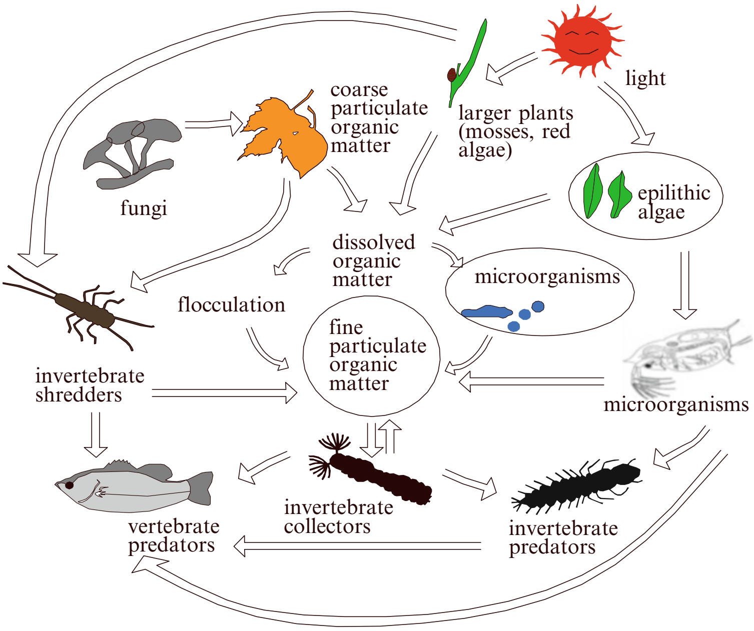


Fig. 3.6. Stream ecosystem and bio-community (after FISRWG 1997).

(invertebrates greater than 0.5 mm in length, such as mayflies, stoneflies, caddis flies, crayfish, worms, clams, and snails), and vertebrates (fish, amphibians, reptiles, and mammals) as shown in Fig. 3.6. 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* sp.; and emergent plants, such as *Phragmites communis* Trin (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 3.7 shows microhabitats with moss on cobbles, submerged macrophytes species *Potamogeton* sp., floating plants species *Lemna minor*, and emergent plants species *Phragmites communis* Trin (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

Fig. 3.7. Aquatic plants: (**a**) moss on cobbles; (**b**) *Potamogeton* sp.*;* (**c**) *Lemna minor*; and (**d**) *Phragmites communis* Trin.

and amphipods). Other invertebrates filter smaller organic material from the water, known as filterers (blackfly larvae, some mayfly nymphs, and some caddis fly larvae); scrape material off the surfaces of bedrock, boulders, and cobles, known as scrapers (snails, limpets, and some caddis fly and mayfly nymphs); or feed on material deposited on the substrate, known as collectors (dipteran larvae and some mayfly nymphs) [24]. Some macroinvertebrates are predatory, known as predators, such as dragonfly, which prey on small vertebrates. Figure 3.8 shows typical species of the five groups with different ecological functions.

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

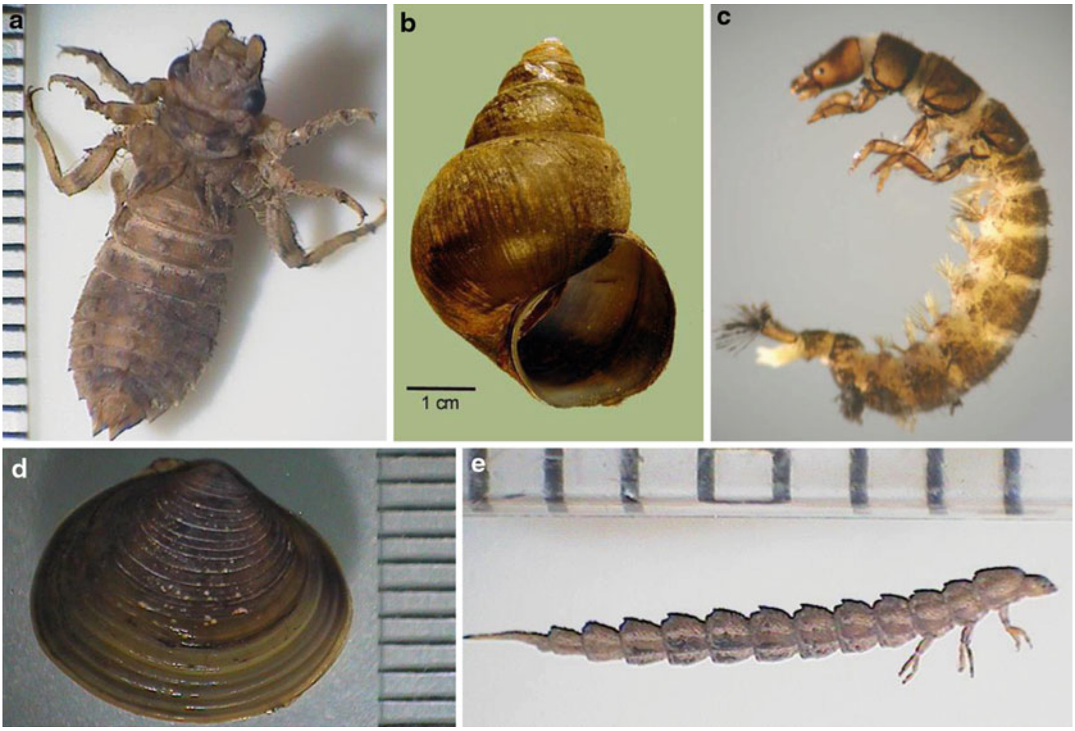


Fig. 3.8. Typical species of the five groups of benthic invertebrates with different functions in the food chain: (**a**) Gomphidae (dragonfly-predator); (**b**) Viviparidae (scraper); (**c**) Hydropsychidae (collector-filterer); (**d**) Corbiculidae (collector-filterer); and (**e**) Haliplidae (shredder).

the stream is small, and environmental fluctuations occur with a greater intensity and frequency; therefore, species richness is lowest [9].

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 [25, 26]. 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, economic, 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 [27].

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.

1.5. 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. Ecological functions 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 [28]. 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 [29].

1.5.1. 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

Fig. 3.9. The Fazi River, an urban stream in Taichung City, provides habitats for benthic invertebrates, fish and birds.

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 Taichung City, as shown in Fig. 3.9. 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 macroinvertebrates, 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. 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.

Fig. 3.10. 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.

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.

1.5.2. Conduit Function

To act as a route for the flow of energy, materials, and organisms is known as the conduit function, as shown in Fig. 3.10. 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 neotropics 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 Fig. 3.10. 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 airsheds. 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.

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.

1.5.3. 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 Fig. 3.11.

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

Fig. 3.11. A stream functions as a boundary of land uses, plant communities, and some less mobile wildlife species.

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 [30].

1.5.4. *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 Fig. 3.12. 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, groundwater, 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: (a) decreased downstream flooding through floodwater moderation and/or uptake, (b) containment of sediments and other materials during flood stage, and (c) source of soil organic matter and waterborne organic matter [31].

Fig. 3.12. 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.

2. ECOLOGICAL STRESSES TO RIVERS

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 hydrologic and hydraulic features of the river, and these changes may cause changes in sediment transportation, habitat, and ecology [32].

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 [33]. 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 °C for the USA and Europe, but 100 mm/month is used as equivalent to 10 °C for a tropical rain forest [33]. In this book, 30 mm/month is equivalent to 10 °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 °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.

A huge landslide may totally destroy the terrestrial and aquatic ecosystem of a river. Figure 3.13 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 m^3. The stream and vegetation on slopes were buried underneath the 180-m-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 3.14a shows the high sediment concentration in a stream in Taiwan, southeast China, which causes a strong stress on the aquatic bio-community. 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 3.14b 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.

Fig. 3.13. Wenjiagou Landslide in Mianzhu City buried the stream and vegetation.

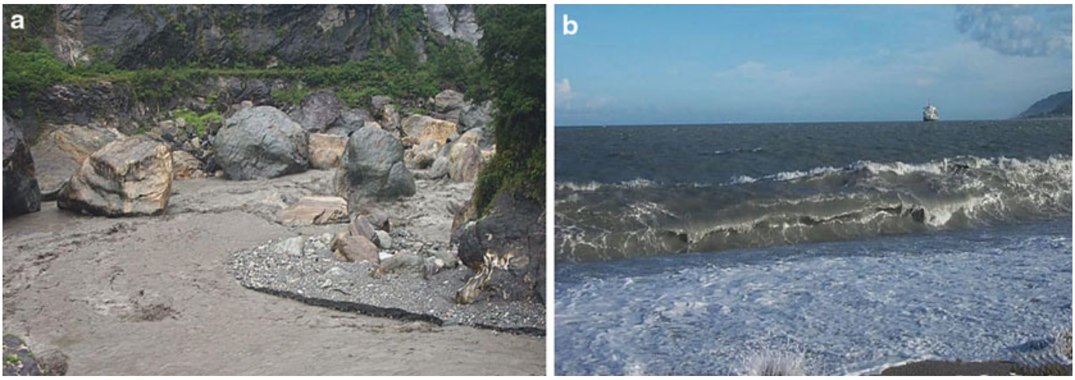


Fig. 3.14. (**a**) High sediment concentration in a stream in Taiwan, southeast China, which results in low transparency, low dissolved oxygen, and sediment coating the substrate; (**b**) Turbid seawater with high concentration of sediment impacts on fish and invertebrate communities.

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 3.15a shows that a couple of beavers skillfully use nature's building materials and construct a wood dam with tree branches on the Spring Pond in Pennsylvania, and Fig. 3.15b 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

Fig. 3.15. (**a**) A couple of beavers began to construct a dam with tree branches on the Spring Pond in Pennsylvania, USA; (**b**) The 3-m-high beaver dam forms a pond, which provides a good habitat for fish and birds.

"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. 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" are 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

[34]. 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.

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 USA. 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.

2.2.1. *Dams*

Ranging from small temporary structures to huge multipurpose structures, human-constructed barriers can have profound and varying impacts on stream corridors. Barriers affect resident and migratory organisms in stream channels. Power plants may kill fish when they swim through the turbines. Figure 3.16a shows that many birds are searching for dead fish at the outlets of a hydropower plant in Korea, which were killed when they swam through the turbine; Fig. 3.16b 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. 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

Fig. 3.16. (**a**) Birds are searching for dead fish at the outlets of a hydropower 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.

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.

2.2.2. Channelization 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. 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.

2.2.3. Fragmentation of Habitat

Some river training works result in the fragmentation and isolation of habitats. Figure 3.17 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

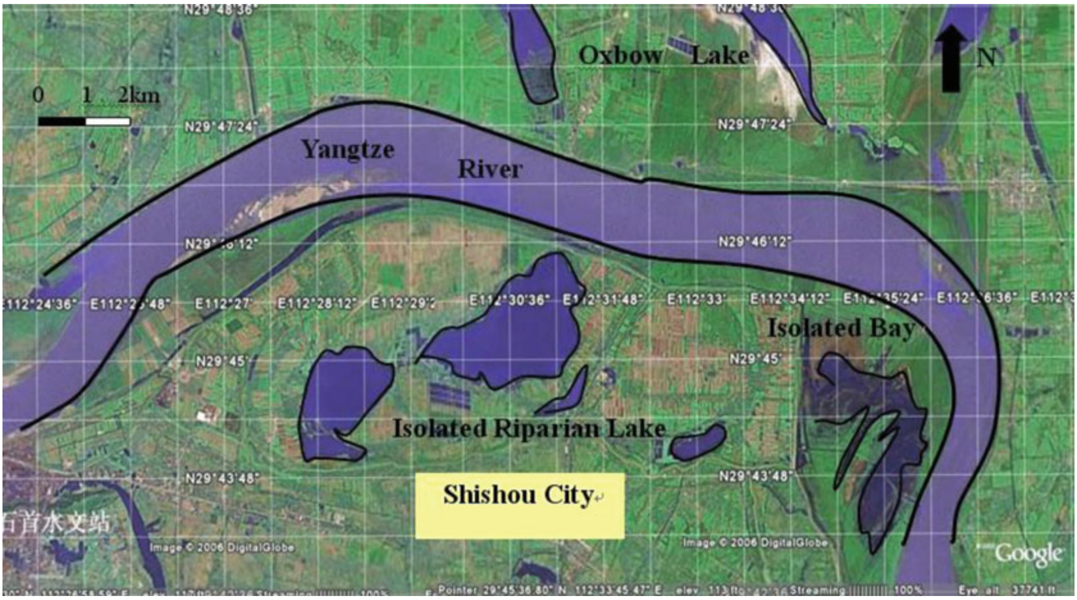


Fig. 3.17. Isolation of riparian lakes along the Yangtze River results in fragmentation of habitat (Satellite image from the web http://earth.google.com).

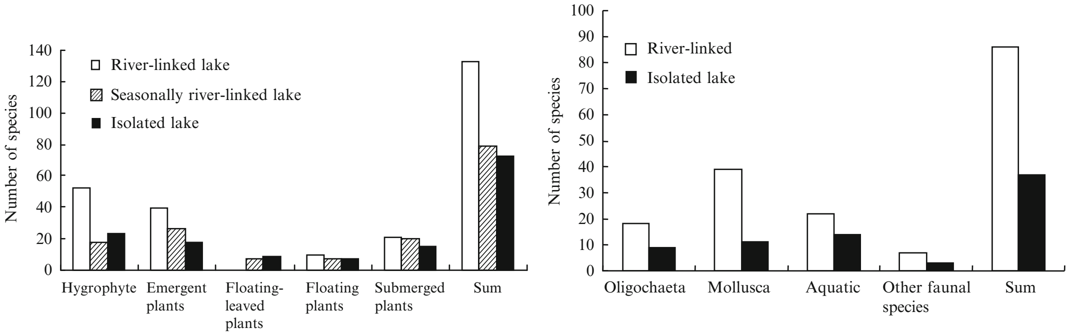


Fig. 3.18. Comparison of species richness of aquatic plants and benthic invertebrates in isolated lakes and river-linked lakes in the middle and lower Yangtze River basin (after Wang and Wang 2008).

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.

Cutoff of riparian lakes from the Yangtze River stressed the complex ecosystem in the lakes and the river. Figure 3.18 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 [35]. The connection of the isolated lakes with the Yangtze River was cut off in

Fig. 3.19. (**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.

the past decades, which has resulted in continuous reduction of species. The cutoff 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.

2.2.4. Mining

Gold placer mining in rivers has become an extreme intensive disturbance to the stream ecology in southwest China. Figure 3.19a 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 widespread. Since the 1980s, gravel mining has become a serious ecological stress in many rivers throughout China, as shown in Fig. 3.19b. Gravel and coarse sand are mined for building materials. Gravel mining causes loss of habitat for benthic bio-communities 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 river 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. 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.

2.2.5. 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. Toxic runoff or precipitates can kill streamside vegetation or can cause a shift to species more tolerant of polluted conditions. 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.

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 bottom-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.

2.2.6. 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 2–16 times, with proportional reductions in groundwater recharge [36]. Figure 3.20 conceptually shows the effects of different amounts of impervious cover on the water balance for a watershed.

2.2.7. 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

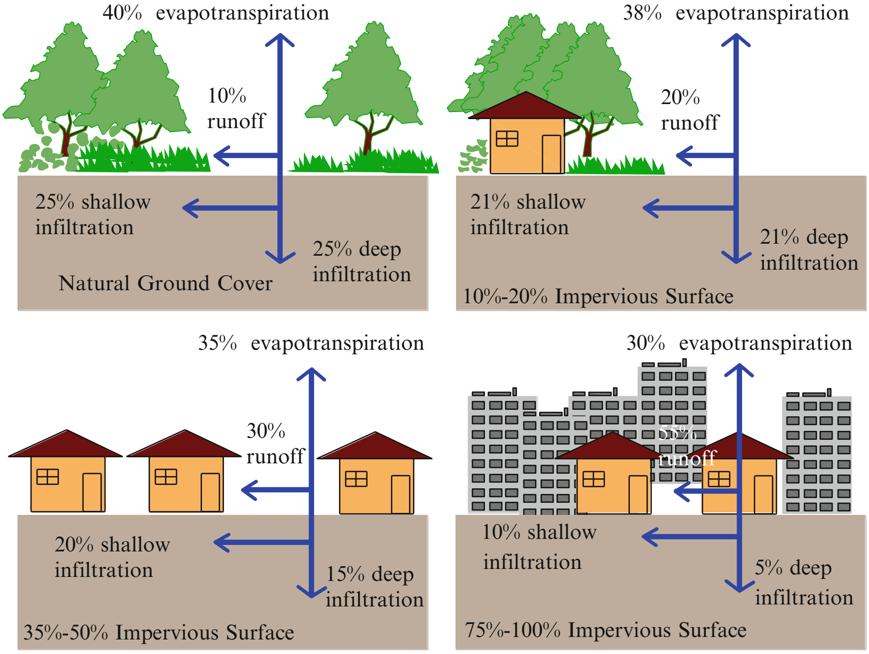


Fig. 3.20. Effects of different amounts of impervious cover on the water balance for a watershed (after FISRWG 1997).

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 groundwater 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.

Fig. 3.21. (**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.

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.

2.2.8. 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 Fig. 3.21a. 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 Fig. 3.21b.

2.2.9. 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 [37]. In areas where the stream can support recreational boating, the system is vulnerable to additional impacts.

2.2.10. Forestry

In addition to the changes in water, sediment, and nutrient 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 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 groundwater 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 niloticus*) to Lake Victoria, in East Africa, causing the extinction of dozens of species of small endemic cichlid fish.

Exotic animals are a common problem in many areas in the USA 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 3.22 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 has imposed an intense stress on native biological communities in the western USA. Without the normal checks and balances found in their native habitat in the eastern USA, bullfrogs reproduce prodigiously and prey on numerous native amphibians, reptiles, fish, and small mammals.

Golden mussel (*Limnoperna fortunei*) is an invasive filterer species of macroinvertebrate. Originally the species comes from south China, which has spread to various regions, including Japan, Australia, Argentina, Thailand, India, Brazil, and Europe [38]. The species colonizes habitats with water temperature between 8 and 35 °C, flow velocity less than 2 m/s, water

Fig. 3.22. *Cambarus clarkaij* has been introduced in many waters in south China, resulting in ecological problems.

Fig. 3.23. Colonization of golden mussel on concrete walls in a water transfer tunnel and attachment of golden mussel on a concrete fragment with high density.

depth less than 10 m with or without sunlight, dissolved oxygen higher than 1.0 mg/L, and pH higher than 6.4 [39–41]. 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 drinking water transfer tunnels and pipelines results in macro-fouling [42] and causes high resistance to water flow and damage to pipeline walls. This along with dead mussels decay harms the surrounding water quality [43, 44]. Figure 3.23 shows colonization of golden mussel in the water transfer tunnels in Shenzhen, southern China, and attachment of 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.

Introduction of exotic species has inevitably occurred worldwide, and this is accelerating following economic and ecological globalization. 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 stream bank.

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*.

Spartina alterniflora was introduced from the USA 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 alterniflora*. The species has over to spread the neighboring coastal areas. Coastal areas and estuaries dominated by reed (*Phragmites communis* Trin) have been colonized and occupied by *Spartina alterniflora*. The fishery harvest has been significantly reduced. Figure 3.24a shows *Spartina alterniflora* 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. 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 3.24b shows the *Eupatorium adenophorum* in Yunnan Province in southwest China.

Eichoimia crassips 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 3.24c shows *Eichoimia crassips* spreading quickly in a polluted stream in Beijing.

Ambrosia artemisia L. and *Ambrosia trifida* 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

Fig. 3.24. (**a**) *Spartina alterniflora* in the Yangtze River estuary; (**b**) *Eupatorium adenophorum* in Yunnan Province in southwest China; (**c**) *Eichoimia crassips* spreading quickly in polluted waters; (**d**) *Ambrosia artemisia* L. in northeast China.

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 3.24d shows the *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 term "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 [45]. 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 or 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 [45].

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 [46]. 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 US 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 [46].

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 [30]:

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 [47]. The selection of indicator species should be done through corroborative research.

3.1.2. Aquatic Macroinvertebrates

Aquatic macroinvertebrates 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 [47]. 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 [48]. Using benthic macroinvertebrates is advantageous for the following reasons: (a) they are good indicators of localized conditions, (b) they integrate the effects of short-term environmental variables, (c) degraded conditions are easily detected, (d) sampling is relatively easy, (e) they are in the middle of the food chain and provide food for many fish of commercial or recreational importance, and (f) macroinvertebrates are generally abundant [49–51].

Field sampling of macroinvertebrates 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 1 m^2 [52]. 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 macroinvertebrates 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 [53], and each species is assigned to an FFG based on the literature [49, 54].

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 [55].

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 as follows: (a) they are good indicators of long-term effects and broad habitat conditions, (b) fish communities represent a variety of trophic levels, (c) fish are at the top of the aquatic food chain and are consumed by humans, (d) fish are relatively easy to identify, and (e) water quality standards are often characterized in terms of fisheries. However, using fish as indicators is inconvenient because (a) the cost of collection is high, (b) long-term monitoring and a large number of samplings are needed to have reliable results and statistical validity may be hard to attain, and (c) the process of sampling may disturb the fish community [56].

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 [57]. The assessment method using fish as indicator has been studied and applied in many large rivers [49].

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 [58]. 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 [59].

In general the advantages of using birds and mammals as indicator species are that (a) they are good indicators of long-term effects and broad habitat conditions, including terrestrial and aquatic ecosystems; (b) they are at the top of the food chain; (c) they are relatively easy to identify; and (d) some restoration projects aim at restoration of endangered birds and mammals. The disadvantages are that (a) the cost of collection is high, (b) long-term monitoring is needed to have reliable results, and (c) they are not sensitive to aquatic habitat conditions (e.g., hydrologic 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 [60]. Rodgers et al. (1979) describe functional measurements of algal communities, such as primary productivity and community respiration, to evaluate the effects of nutrient enrichment [61].

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 index [62] and the most important characteristic of biodiversity:

$$S = \text{total number of species in the samples from a sampling site.} \tag{3.1}$$

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 identify 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 [63].

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 [64]:

$$S = cA^z \tag{3.2}$$

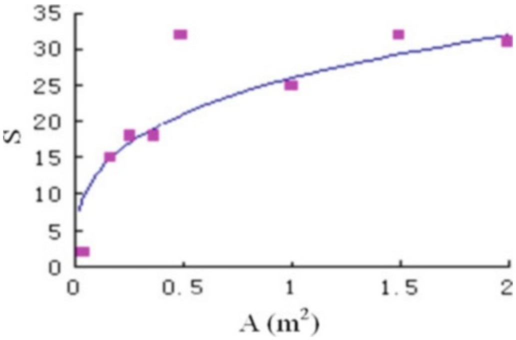


Fig. 3.25. Relation between the species richness in a sample and the sampling area at each site.

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 macroinvertebrates. 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 [64]:

$$S = 10A^{0.24} \tag{3.3}$$

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 macroinvertebrate community is more localized and needs much less sampling area for assessment of local ecosystems. If a river ecosystem with high heterogeneity of habitat is assessed with macroinvertebrates 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 workload increases with the sampling area; therefore, ecologists prefer small sampling areas as long as a sufficient number of species can be sampled. Figure 3.25 shows the relation between the number of species in a sample and the sampling area at each site [52]. 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 [64]:

$$\frac{\mathrm{d}N}{\mathrm{d}t} = rN\left(1 - \frac{N}{K}\right) \tag{3.4}$$

in which r represents the intrinsic exponential growth rate of the population when its size is very small (i.e., 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, $\mathrm{d}N/\mathrm{d}t$ 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

$$N = \frac{K}{1 + \frac{K - N_0}{N_0}\mathrm{e}^{-rt}} \tag{3.5}$$

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 Dongting Lake, or for a bio-community, e.g., benthic macroinvertebrates 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

$$D = \left[\sum_{i=1}^{S}\left(\frac{{n_i}^2}{N^2}\right)\right]^{-1} \tag{3.6}$$

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 [65]:

$$M = (S - 1)/\log_{\mathrm{e}}^{N} \tag{3.7}$$

The Shannon-Weaver index, developed from information theory and integrating the species richness and evenness of the abundance distribution, is given [66]:

$$H = -\sum_{i=1}^{S} \frac{n_i}{N} \ln \frac{n_i}{N} \tag{3.8}$$

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. Equation (3.8) 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 [67]:

$$B = H \ln N = -\ln N \sum_{i=1}^{S} \frac{n_i}{N} \ln \frac{n_i}{N} \tag{3.9}$$

Macroinvertebrates census data from nine sites along the East River in south China can be used to illustrate these different methods of presentation, as listed in Table 3.2 [68]. 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 3.26 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 impacts 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.

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 biodiversity and bio-community.

Table 3.2
Species of benthic macroinvertebrates 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, *Corbicula 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, *Corbicula nitens* (1); Noteridae (1); *Whitmania* (1); *Hirudinea* sp. (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)

Biodiversity of macroinvertebrates in different types of abandoned channels was different, which is also due to different riverbed habitats. There are four types of abandoned channels: old river courses, oxbow lakes, oxtail lakes, and riparian wetlands, which result from avulsions, meander cutoffs, ice-jam floods, and stem-channel shifts, respectively. These posterior three types belong to freshwater ecosystems. Oxbow lakes result from the natural cutoff of meanders. In meandering rivers, a continuing increase in the amplitude and tightness of bends may result in a threshold sinuosity at which the river can no longer maintain its shape and a cutoff develops. Oxbow lakes may also result from artificial cutoffs. In general, artificial cutoffs cause intensive erosion in the new channel in the first several years of formation. The new channel is not stable during the intensive fluvial process. Oxtail lakes are generated from the fluvial process of anastomosing rivers. In northeast China, some rivers flow from south to north. When the northern section of the river freezes, the water will

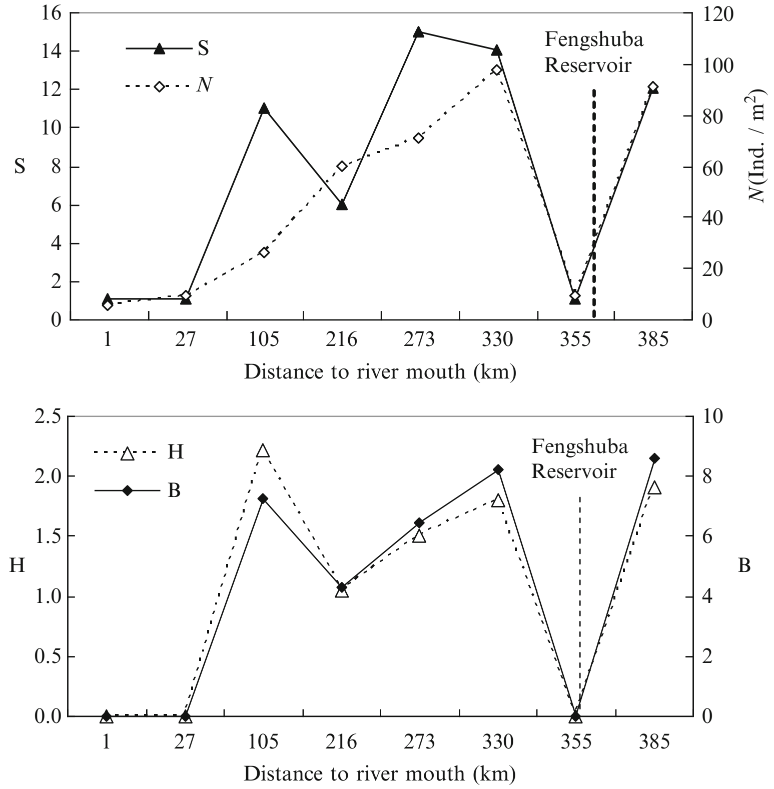


Fig. 3.26. Species richness, *S*; number density of individual invertebrates per area, *N*; Shannon-Weaver index, *H*; and the bio-community index, *B*, as functions of distance to the river mouth.

still flow upstream to the north. This forms anastomosing rivers, which are not very stable. If one of the parallel channels is scoured deeper than the others, all available water may flow into this channel and abandon the others. The lower part of the abandoned channel remains connected with the main channel and becomes a channel-shaped lake. These lakes are different from the oxbow lakes in shape and origin and are named as such because they look like oxtails. Many riparian wetlands result from the shifting of channels. Sediment is then deposited in wide river sections and forms bars. Under some circumstances, one channel of braided river develops into the main channel, and the other channels and the bars become a wetland.

Field investigations were carried out in the Yellow River, Songhua River basin, and East River. The locations of the study areas and sampling sites are shown in Fig. 3.27. Environmental conditions of the sampling sites and macroinvertebrate biodiversity are given in Table 3.3. It can be found that East River that is freely connected with the mainstream was characterized by the highest biodiversity. Species diversity can also be assessed using K-dominant curves, which combines the two aspects of diversity-species richness and evenness. Using this method, dominance patterns can be represented by plotting the accumulative

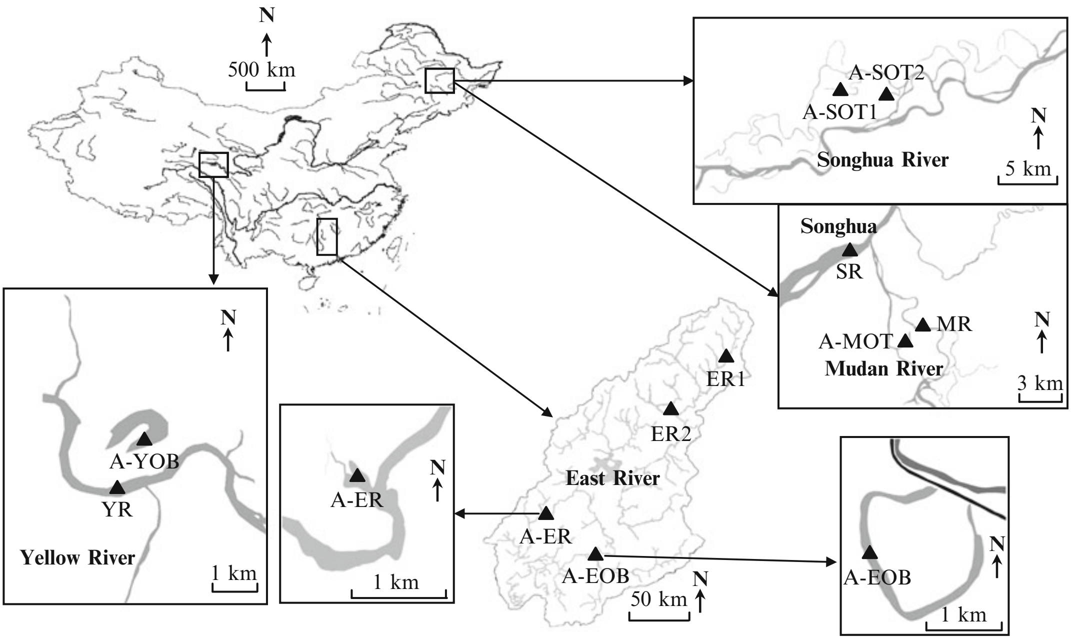


Fig. 3.27. Locations of study areas and sampling sites. The site abbreviations are the same as defined in Table 3.3. *Filled triangle* represents the sampling site.

abundance of each species (%) ranked in decreasing order of dominance. The lower curve indicates even individual distribution and higher biodiversity. Figure 3.28 shows K-dominant curves of macroinvertebrates in abandoned channels and their adjoining rivers. Figure 3.28 reports similar results as shown in Table 3.3. The different biodiversity of these channels was ascribed to different habitat conditions.

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 [69]. When measuring diversity, it is important to clearly define the biological objectives, stating exactly what attributes of the system are of concern and why [70]. 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.

Table 3.3
Environmental conditions of the sampling sites and macroinvertebrates biodiversity

Site		Code	Location	Connection frequency (/year)	Substrate	Velocity (m/s)	Water depth (m)	*H*	*B*
Yellow River	Oxbow lake of the Yellow River at Kesheng	A-YOB	N 34°12′4″ E 101°33′40″	0.3	Clay and silt, dense emerged and submerged plants (cover: 1/3)	0.0–0.3	0.3–1.0	1.74	11.86
	Mainstream of the Yellow River at Kesheng	YR	N 34°12′3″ E 101°33′39″		Silt, sand and cobbles	0.1–1.0	0.1–1.0	2.08	9.86
Songhua River	The first oxtail lake of the Songhua River in Harbin	A-SOT1	N 45°47′1″ E 126°23′25″	1.0	Silt and fine sand, aquatic plant (cover: 1/3)	0.0–0.2	0.1–1.5	1.45	7.58
	The second oxtail lake of the Songhua River at Wanbao	A-SOT2	N 45°47′58″ E 126°32′18″	0.1	Silt and fine sand	0.0–0.2	0.1–1.5	0.71	4.26
	Mainstream of the Songhua River at Yilan	SR	N 45°47′5″ E 126°36′37″		Sand and gravel	0.1–0.8	0.1–3.0	0.93	4.08
Mudan River	Oxtail lake of the Mudanjiang River at Xiaowujia	A-MOT	N 45°49′5″ E 126°44′4″	1.0	Clay and silt	0.0–0.2	0.1–0.6	1.42	9.93
	Mainstream of the Mudanjiang River at Xiaowujia	MR	N 45°49′4″ E 126°43′59″		Sand and gravel	0.3–0.8	0.1–1.5	1.68	3.68
East River	Oxbow lake of the East River	A-EOB	N 23°3′23″ E 114°25′34″	0.0	Fine sand	0.0	0.0–3.0	0.89	2.76
	Zengjiang Bay (riparian wetland of the East River)	A-ERW	N 23°27′1″ E 113°54′8″	Always connected	Clay, silt and sand, dense submerged plants (cover: 1/2)	0.0–0.5	0.0–3.0	2.57	15.02
	The first site in the mainstream of the East River at Shangpingshui	ER1	N 24°34′15″ E 115°29′46″		Cobbles and boulders, aquatic plants (cover: 1/4)	0.2–1.0	0.1–0.5	1.97	8.92
	The second site in the mainstream of the East River at Yidu	ER2	N 24°17′36″ E 115°7′49″		Cobbles	0.2–1.5	0.2–1.0	1.66	7.41

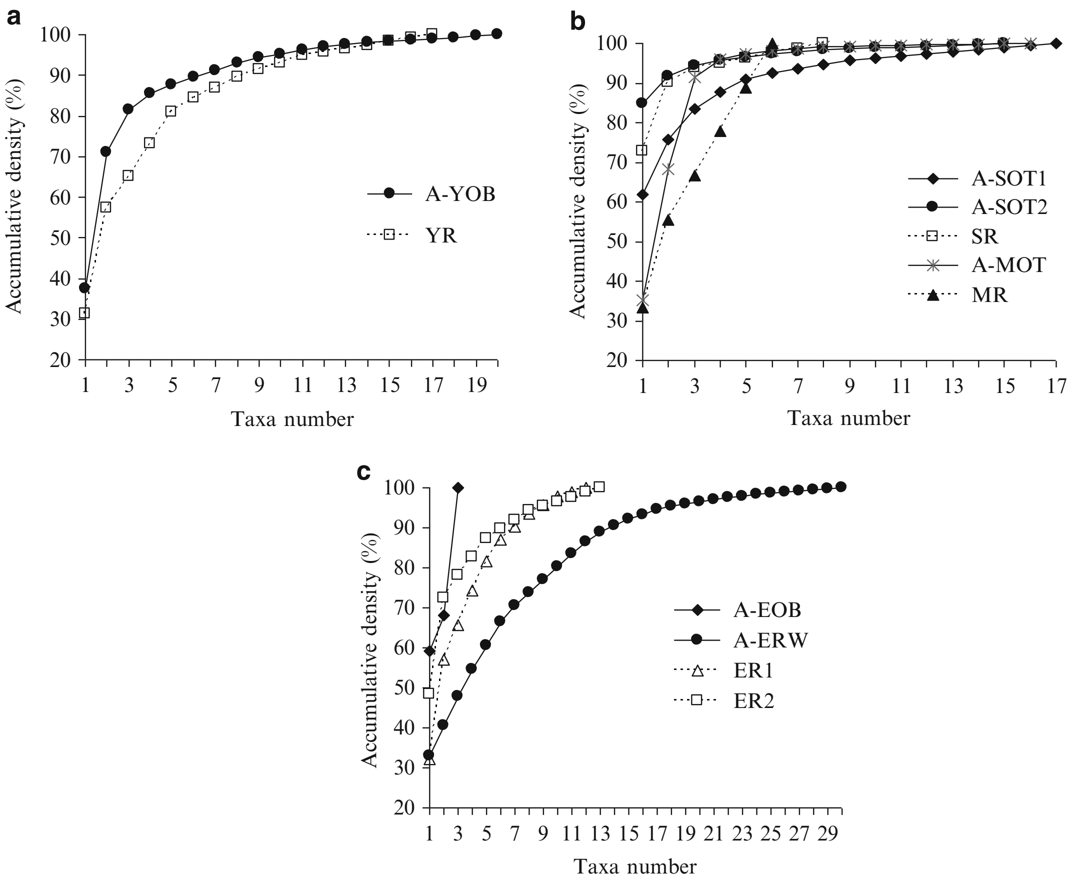


Fig. 3.28. K-dominant curves of macroinvertebrates in abandoned channels (*solid line*) and their adjoining rivers (*dashed line*). The site abbreviations are the same as defined in Table 3.3.

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 [71]. 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:

$$\beta = \frac{M}{\frac{1}{S}\sum_{i=1}^{S} m_i} \tag{3.10}$$

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 macroinvertebrates 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 3.29 shows the location of 8 sampling sites. Samples were taken from five 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 macroinvertebrates 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 subsampling areas in order to reflect diversified ecological conditions. After sampling, macroinvertebrates and associated material were immediately preserved in ethanol and were subsequently processed and identified in the laboratory. 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 [72]. At the site ① the 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 an abandoned channel of the Yellow River and has been cut off 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

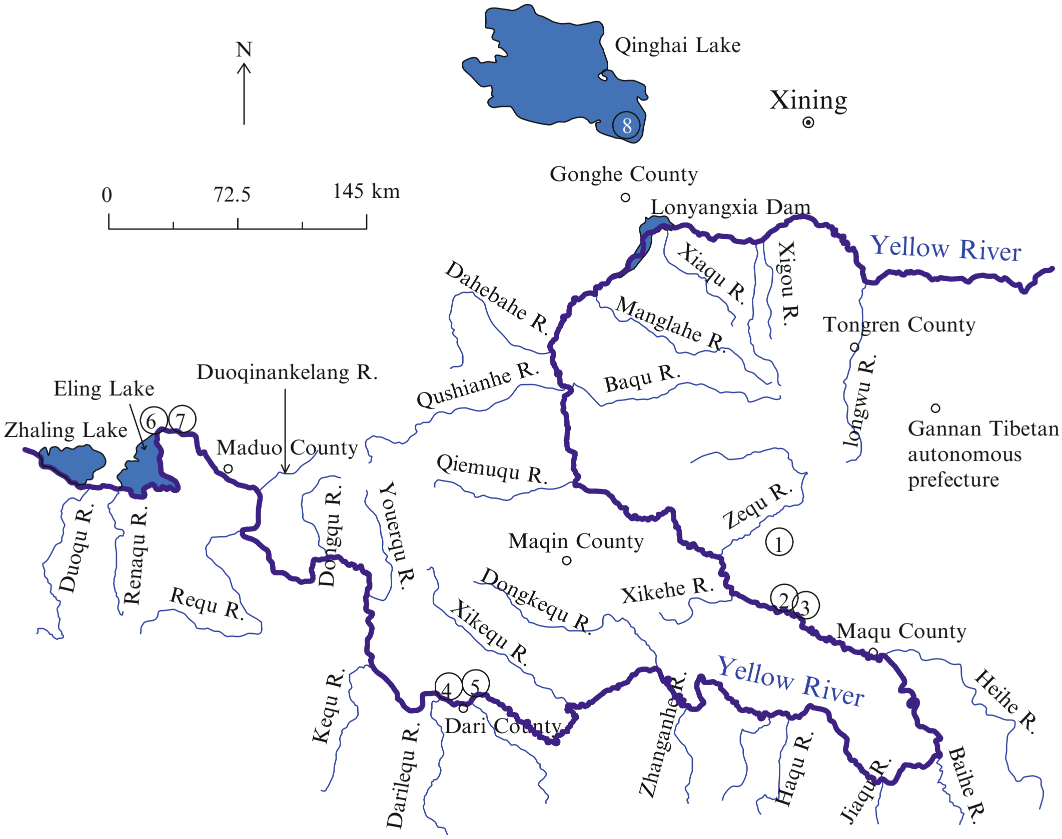


Fig. 3.29. Location of the 8 sampling sites in the source region of the Yellow River.

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 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 3.4 lists the species of macroinvertebrates identified from the samples of each site with the number density (ind/m^2) of each species in the parentheses. 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 macroinvertebrates belonging to 24 families and 44 genera were identified. The average density and wet biomass of macroinvertebrates in the eight sampling stations were 360 ind/m^2 and 2.3934 g/m^2, 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 3.30 shows the representative species of macroinvertebrate in the sampling sites, which are dominant species or typical species at each site.

Table 3.4
Species composition of macroinvertebrates 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

The taxa richness *S* and the index *B* at each site are not high. In other words, the biodiversity of the sampling sites is 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 is quite different from the river and bay. These riparian waters are important in aquatic biodiversity.

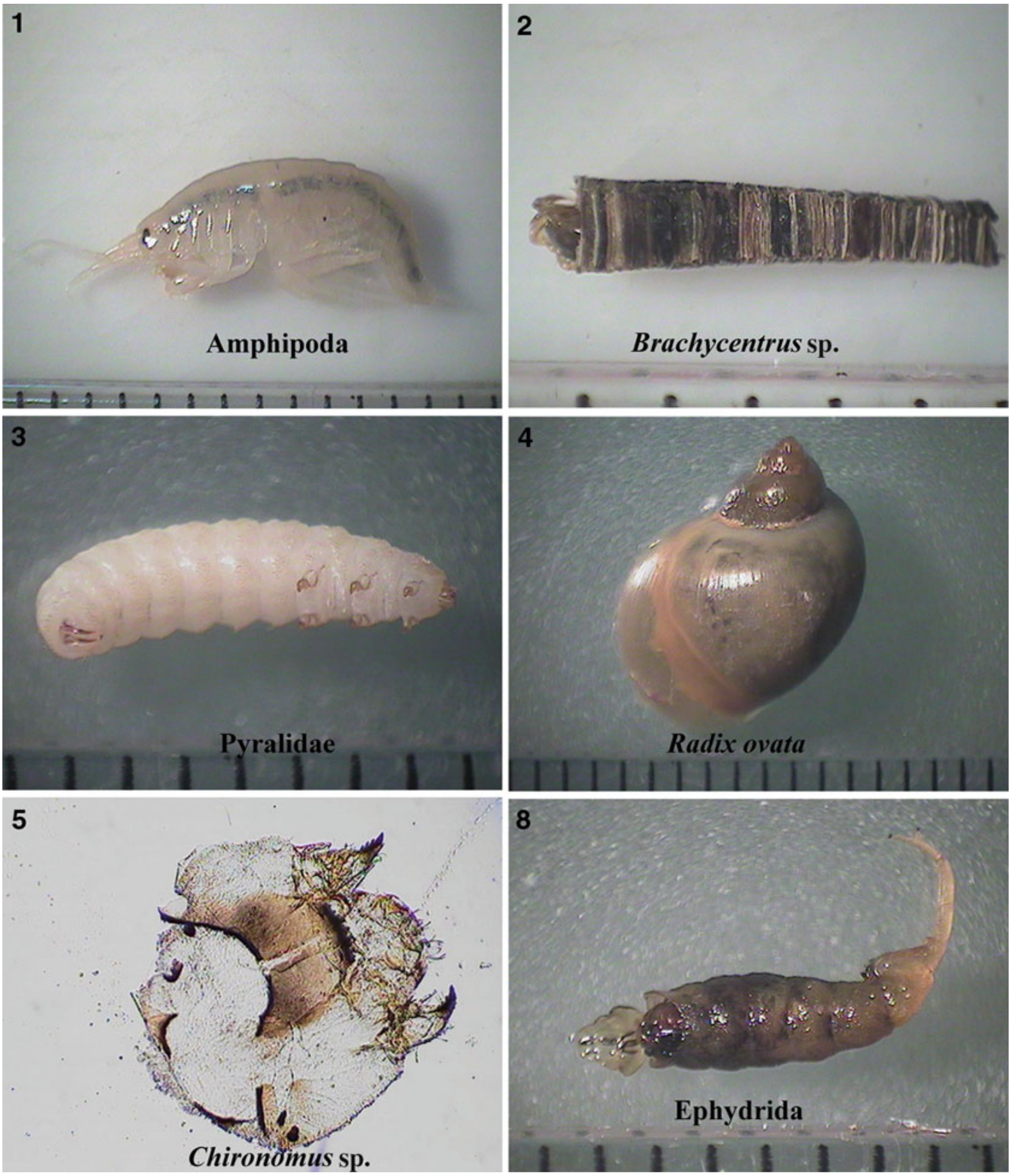


Fig. 3.30. Representative species of macroinvertebrate from the sites 1, 2, 3, 4, 5, and 8.

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 (3.10). 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. Samples of benthic invertebrates were taken from eight 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.

3.2.4. Indices of Biotic Integrity

Karr's IBI

Fish represent the top of the aquatic food chain, and, thus, the quality and composition of the fish community comprise 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 [73]: species richness and condition, indicator species, trophic function, reproduction function, and individual abundance and condition.

Considering the foregoing considerations, Karr (1981) proposed and revised the Index of Biotic Integrity (IBI) to evaluate stream quality at the fish community level [74]. The Karr's IBI is comprised of 12 metrics to define fish community structure. 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 stream types [75]. As recommended by Karr et al. [57], IBI metrics require adjustment for the region to which the index is applied. The basic components of Karr's index are listed in Table 3.5. It is recognized that stream size is an important factor when refining the IBI to a geographic region.

The definitions of the 12 metrics are described as follows [57, 76]:

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

Table 3.5
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 anomalies (DELT)	0–2 %	2–5 %	>5 %

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 (*Ambloplites rupestris*) 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 USA, 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 USA, 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 crustacea (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

Table 3.6
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

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 [76]. The results of the reach sampling are combined to define a sampling site.

IBI Examples

Karr's IBI concept has been adapted and modified from its Midwestern USA 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 USA [77].

The IBI developed for Taiwan [78] is an example, where the majority of Karr's original metrics (with slight modifications) were applied, but the scoring criteria were modified (Table 3.6). 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

Table 3.7
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 anomalies (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 %

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.

Karr's IBI and its many regional modifications for areas throughout the USA and around the world have generally been well calibrated to small "wadable" streams, but applications in larger rivers are less common [79]. 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 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 number-based metrics indicate the diversity of pathways that energy could follow and the potential for intra- and interspecific interactions [79].

The large river IBI for southern Wisconsin is listed in Table 3.7. Definitions of some of the "new" metrics are given as follows [76, 79]:

Weight per unit effort—Weight (biomass) to the nearest 0.1 kg of fish collected per 1,600 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, and 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 [79]. The fact that the IBI metrics in Table 3.7 reflect conditions on the Mississippi River and Ohio River indicates that these metrics might be a good beginning point for developing IBIs for the other large rivers of the world.

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 [79]. 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 [80–83]. 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. Bioassessment

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 [30]. Bioassessment often refers to use of biotic indices or composite analyses, such as those used by the Ohio Environmental Protection Agency [84], and rapid bioassessment protocols (RBP), such as those documented

Table 3.8
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 h per site (no standardized sampling)	Order, family/ field	One highly trained biologist
II	Benthic invertebrates	Intermediate; 1.5–2.5 h per site (all taxonomy performed in the field)	Family/field	One highly trained biologist and one technician
III	Benthic invertebrates	Most rigorous; 3–5 h per site (2–3 h of total are for lab taxonomy)	Genus or species/laboratory	One highly trained biologist and one technician
V	Fish	Low; 1–3 h per site (no fieldwork involved)	Not applicable	One highly trained biologist
VI	Fish	Most rigorous; 2–7 h per site (1–2 h are for data analysis)	Species/field	One highly trained biologist and 1–2 technicians

by Plafkin et al. [49]. 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 macroinvertebrate community. The rapid bioassessment protocols established by the US Environmental Protection Agency were developed to provide states with the technical information necessary for conducting cost-effective biological assessments [49]. The RBP are divided into five sets of protocols, three for macroinvertebrates and two for fish, as shown in Table 3.8.

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. 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: (a) species richness, (b) Hilsenhoff biotic index [85], (c) ratio of scrapers to filtering collectors, (d) ratio of Ephemeroptera/Plecoptera/Trichoptera (EPT) and chironomid abundances, (e) percent contribution of dominant taxa, (f) EPT index, (g) community similarity index, and (h) 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 [86]. 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 [30]:

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 trade-offs. 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 bioassessment. Habitat has a definable carrying capacity, or suitability, to support or produce wildlife populations [87]. 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 [88]. 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 [67]:

$$H_{\mathrm{D}} = N_{\mathrm{h}} N_{\mathrm{v}} \sum_{i} \alpha_i \tag{3.11}$$

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 and 0.5 m. If a stream has three water areas, (a) shallow water, in which the water depth is in the range of 0–0.1 m; (b) mid-depth water, in which the water depth is in the range of 0.1–0.5 m; and (c) 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, $N_{\mathrm{h}} = 3$. 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, $N_{\mathrm{h}} = 2$. 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 and 1 m/s. If a stream has three water areas, (a) lentic area, in which the flow velocity is smaller than 0.3 m/s; (b) mid-velocity area, in which the flow velocity is in the range of 0.3–1 m/s; and (c) 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, $N_{\mathrm{v}} = 3$. If a stream has only lentic and mid-velocity areas, and each of them is larger than 10 % of stream water, $N_{\mathrm{v}} = 2$. The value of N_{v} for other cases can be analogously obtained.

The selection of the critical values of water depth and velocity is determined by studying the habits of species, mainly of macroinvertebrates. It is found from field investigations that in the Yangtze River basin some species in the water depth between 0.1 and 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 [89].

Streambeds consisting of cobbles and boulders are very stable and provide the benthic macroinvertebrates 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 macroinvertebrates 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 3.9. It is well known that large woody debris can substantially contribute to habitat quality in streams [90, 91], 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 3.9.

If a part of the streambed consists of one substrate and another part consists of 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 [88]. 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. 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.

Table 3.9
Substrate diversity, α, values for different substrates (after Wang et al. 2008a)

Substrate	Boulders and cobbles (D>200 mm)	Aquatic grass	Gravel (2–200 mm)	Fluid clay mud ($D < 0.02$ mm)	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

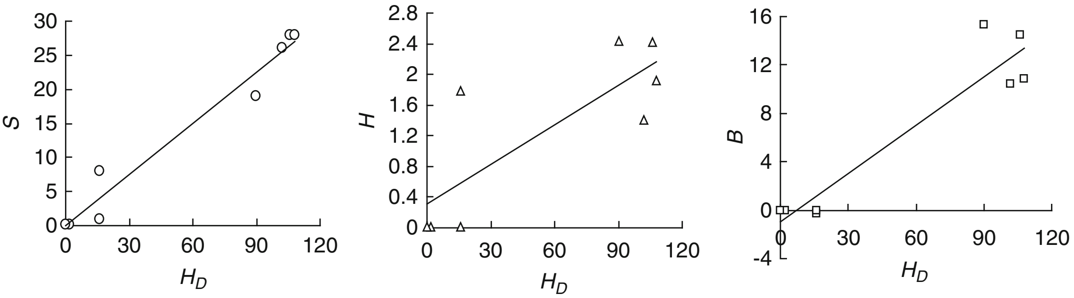


Fig. 3.31. Species richness, S; Shannon-Weaver index, H; and the bio-community index, B, as functions of the habitat diversity index, H_D.

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 macroinvertebrates in several mountain streams in the Xiaojiang River basin in Yunnan Province in southwestern China. Figure 3.31 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 the habitat diversity, the higher are the species richness, the biodiversity, 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-H curve are 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 3.32 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.

High habitat diversity means various habitat conditions. Certainly, influencing variables of macroinvertebrate communities are different under different river habitat conditions. Three Chinese rivers (the Songhua River, the Yongding River, and the West River) with different

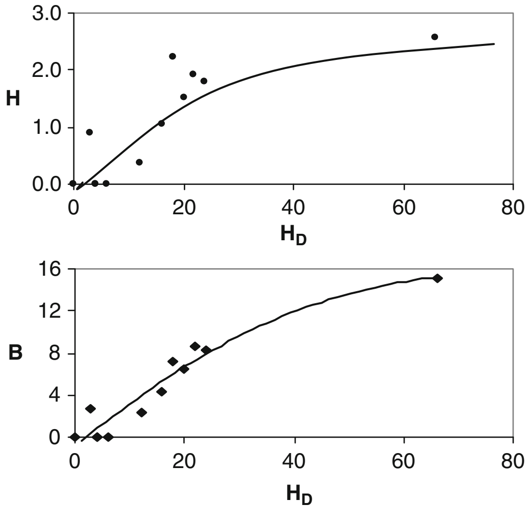


Fig. 3.32. Relation between habitat diversity, H_D, and Shannon-Weaver index, H (*upper*), and the relation between habitat diversity, H_D, and bio-community index, B (*lower*).

latitudes were surveyed May–August of 2009 (high water level) and September–December of 2009 (low water level). Based on canonical correspondence analysis (CCA), water physico-chemical variables (total phosphorus and conductivity) played a key role in structuring macroinvertebrate assemblages in silt substrate, while hydrologic variables (median grain size of substrate and water velocity) mainly affected macroinvertebrate assemblages in stone substrate (Fig. 3.33).

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 [92]; the US Fish and Wildlife Service [93] 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 [94].

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

$$\mathrm{HU} = \mathrm{AREA} \times \mathrm{HSI} \tag{3.12}$$

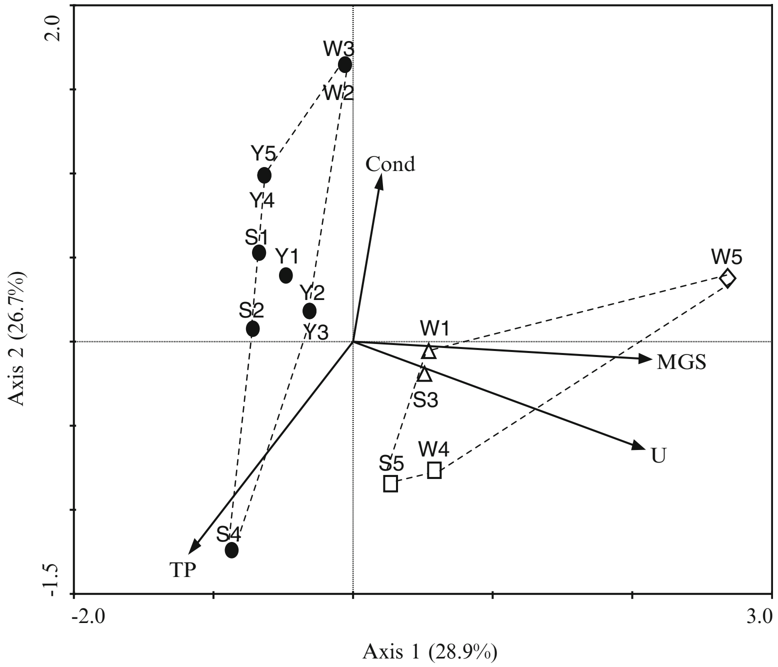


Fig. 3.33. CCA biplots of sites/environments. Major environmental variables influencing abundance and distribution of macroinvertebrates are presented. Environmental variables: *TP* total phosphorus concentration of water (mg/m^3), *MGS* median grain size (mm), *U* water velocity (m/s); *Cond* conductivity (μS/cm). Abbreviated river codes: *S* Songhua River, *Y* Yongding River, *W* West River. Substrate types: *filled circle*, silt; *open triangle*, gravel; *open square*, cobble; *open diamond*, bedrock.

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 US Fish and Wildlife Service primarily for instream flow analysis [95]. 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 life stage-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 [96, 97]. 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 [98, 99]. 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 [100]. The life cycle of the Chinese sturgeon in the Yangtze River mainly comprises spawning, hatching, and growth of 1-year juvenile sturgeon. Brood fish seek suitable spawning sites; fertilized eggs adhere to stone and hatch after about 120–150 h. Whelp sturgeons 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: (a) water temperatures for adults and juveniles (V_1, °C), (b) water depth for adults (V_2, m), (c) substrate for adults (V_3), (d) water temperature for spawning (V_4, °C), (e) water depth for spawning (V_5, m), (f) substrate for spawning and hatching (V_6), (g) water temperature during hatching (V_7, °C), (h) flow velocity during spawning (V_8, m/s), (i) suspended sediment concentration during spawning (V_9, mg/l), and (j) 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:

$$\mathrm{HSI} = \min\left(C_{\mathrm{Ad}},\ \ C_{\mathrm{Sp}},\ \ \ C_{\mathrm{Ha}}\right) \tag{3.13}$$

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

$$C_{\mathrm{Ad}} = \min(V_1,\ \ V_2,\ \ V_3) \tag{3.14}$$

C_{Sp} represents the suitability for spawning

$$C_{\mathrm{Sp}} = \min(V_4,\ \ \ V_5,\ \ V_6) \tag{3.15}$$

C_{Ha} represents the suitability for hatching

$$C_{\mathrm{Ha}} = V_{10} \bullet \min(V_6,\ \ V_7,\ \ V_8,\ \ V_9) \tag{3.16}$$

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 riverbed. Excessive suspended sediment concentration may cause sturgeon eggs to debond, which then affects fertilization and hatching. According to Chang [101], 90 % of sturgeon eggs suffer predation. The data sources used to develop the SIS are listed in Table 3.10, and the *SI* curves are shown in Fig. 3.34. 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 3.10
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 and 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 [102]. Research results indicate that the Chinese sturgeon grows well under a wide range of temperatures; feeding has been recorded from 8 to 29.1 °C [103]. Yan (2003) found that Chinese sturgeons 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 [104]. The optimum temperature for juvenile sturgeon is 22–25 °C [105]
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 to 30 m; 11 Chinese sturgeons detected in the Yanzhiba to Gulaobei reach were distributed at depths from 9 to 19 m [105]
V_3	Substrate (adults)	Juvenile and adult Chinese sturgeons have similar substrate choices as with shortnose sturgeon in the USA. Experiments show that juvenile shortnose sturgeons prefer habitat in sand-mud substrate or gravel substrate [106]. Chinese sturgeons prefer to cruise along river channels with deep trenches and sandy dunes and are fond of resting in pools, backwaters, and places varied terrain [103]
V_4	Water temperature (spawning)	The spawning temperature for sturgeon is 17.0–20.0 °C; spawning will stop when temperature <16.5 °C [107]. 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 [105]. Spawning occurs when temperature is 15.3–20.5 °C; the suitable range is 17.0–20.0 °C, and the optimum is 18.0–20.0 °C [108]
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 Gulaobei, with 10–15-m water depth [103]. The "stable spawning site of Chinese sturgeon" determined by Deng et al. (1991) has a water depth in a range from 4 to 10 m [109]

(*Continued*)

Table 3.10 (Continued)

Variables	Eco-factors	Results of previous research
V_6	Substrate (spawning and hatching)	Gravel and pebbles are present in Chinese sturgeon spawning sites of [110]. The substrate of new spawning sites is composed of sand, gravel with sand, gravel, and stone and gradually coarsens from left to right bank [107]. The substrate of the original centralized spawning sites of Chinese sturgeon was mainly composed of stones and gravels [111]
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 [112]. Water temperature for cultivating fries should be between 12 and 29 °C; the most suitable temperature is 16–24 °C [113]
V_8	Flow velocity (spawning)	Sturgeons 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 [114]. The surface flow velocity at spawning areas is 1.1–1.7 m/s [110]. The flow velocity of spawning areas during spawning season ranges from 0.82 to 2.01 m/s; 57.69 % of fish are distributed between 1.2 and 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 [105]. According to 31 records from 1983 to 2000, the average flow velocity on spawning day was between 0.81 and 1.98 m/s, and 81 % took place in the range of 1.00–1.66 m/s [108]
V_9	Suspended sediment concentration (spawning)	The suspended sediment concentration in reaches downstream from the Gezhouba Dam is between 0.073 and 1.290 kg/m^3, with an average of 0.508 kg/m^3. About 66.67 % of fish are distributed between 0.3 and 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 and 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 and 1.02 kg/m^3, with an average of 0.61 kg/m^3 [107]. The suitable range of suspended sediment concentration for Chinese sturgeon is 0.10–1.32 kg/m^3. From 1983 to 2000, 15 of 31 spawning events were in the range of 0.2–0.3 kg/m^3 [108]

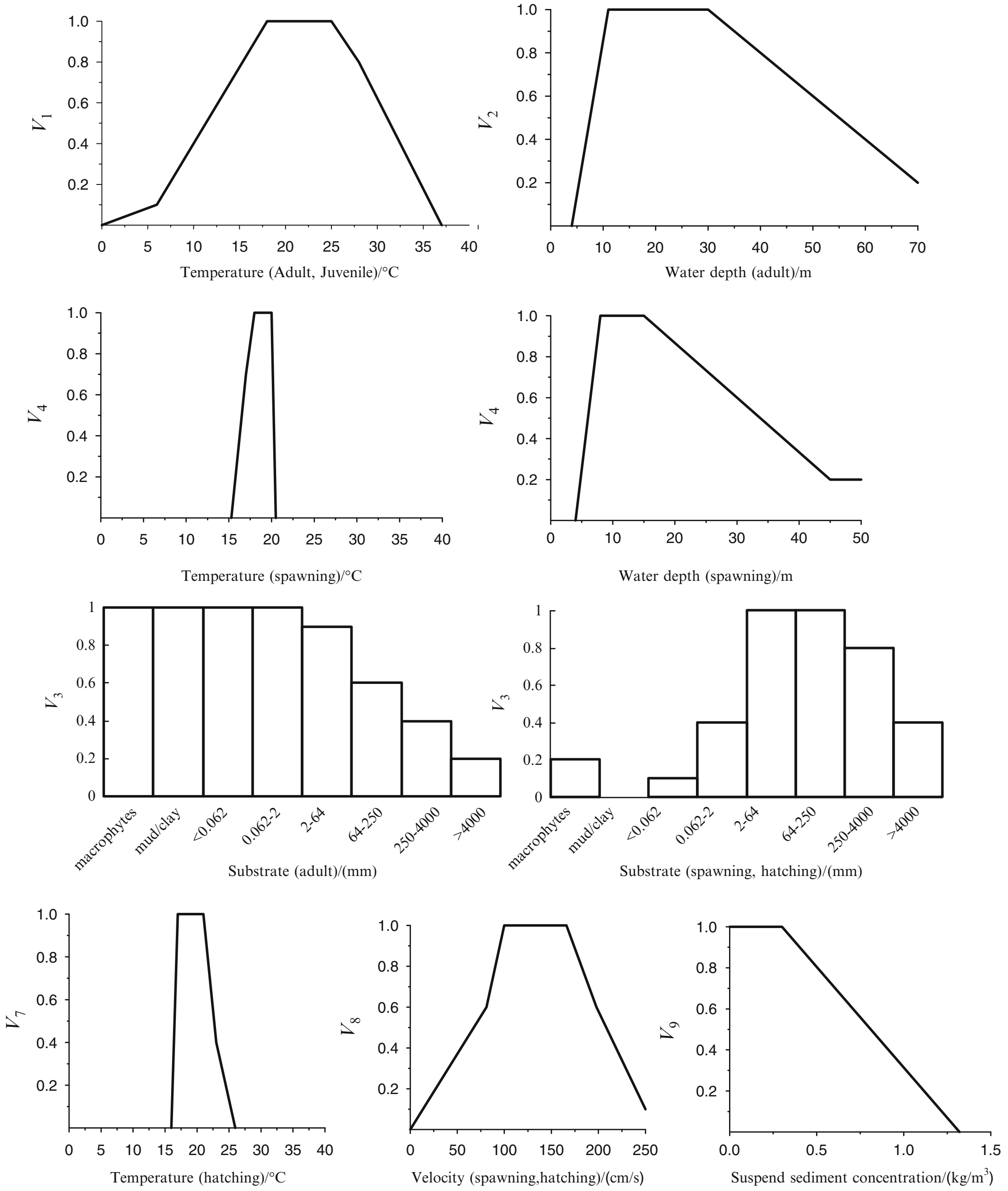


Fig. 3.34. Suitability Index curves for habitat of Chinese sturgeon.

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 frequency of inundation and is a powerful determinant of what plants are likely to be found in various positions in the riparian zone, as

Fig. 3.35. Soil moisture conditions determine the plant communities in riparian areas of the Nile River in Sudan.

shown in Fig. 3.35. 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. 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 [30]:

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 stream flow 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 versus 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 stream flow 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 USA 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, and 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 long-term 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 [115] and the multivolume Impact of Water Level Changes on Woody Riparian and Wetland Communities [116, 117]. 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 flood-prone 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.

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