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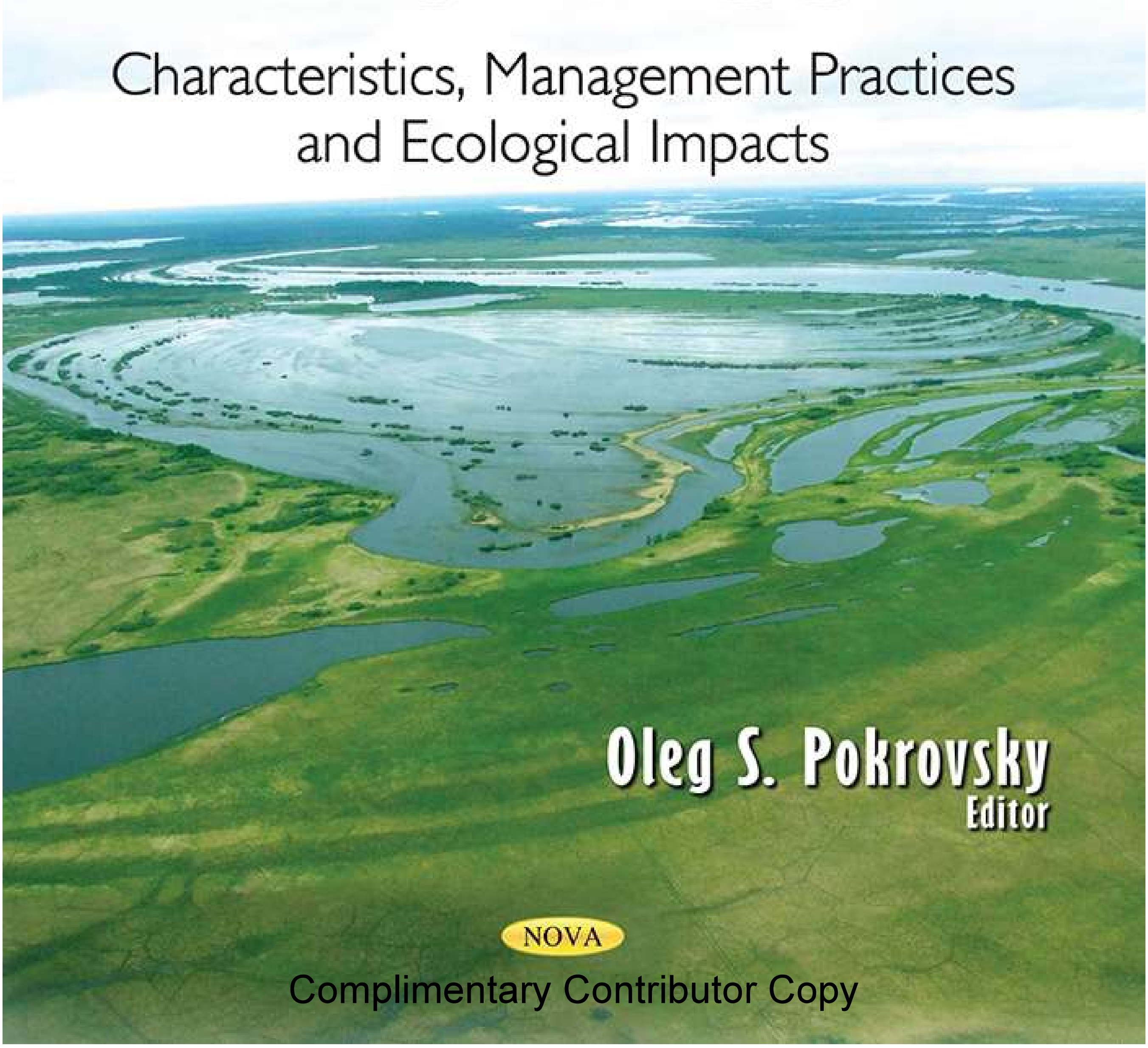
Riparian Zones

Characteristics, Management Practices and Ecological Impacts



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Chapter 1

BIOLOGICAL DIVERSITY AND CURRENT THREATS OF LOTIC ECOSYSTEMS

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ABSTRACT

Although 100,000 out of approximately 1.75 million species described by scientists live in fresh water, knowledge of the total diversity of fresh waters is woefully incomplete, particularly among invertebrates and microbes, and especially in tropical latitudes that support most of the world's species. If trends in human demands for water remain unaltered, and species losses continue at current rates, the probability to conserve much of the remaining biodiversity in fresh water will be very low. Lotic ecosystems are increasingly impacted by multiple stressors that lead to a loss of sensitive species and an overall reduction in diversity. Global environmental change (such as climate) occurring at the global scale, together with nitrogen deposition and runoff, are superimposed upon all of the threats.

In this chapter, I conduct an extensive review of published studies that have qualitatively and quantitatively examined the species richness and the current major threats of a special component of freshwater systems: the lotic ecosystems.

Keywords: biodiversity, rivers, lotic systems, land use, climate change, riparian zones

1. INTRODUCTION

Fresh water consists of only 0.01% of the world's water and approximately 0.8% of the Earth's surface (Gleick, 1996), yet this tiny fraction of global water supports at least 100,000 species (Figure 1) out of approximately 1.8 million, which is almost the 6% of all described species (Dugeon et al. 2006). Inland waters and freshwater biodiversity constitute a valuable natural resource, in economic, cultural, aesthetic, scientific and educational terms. Although their conservation and management are critical, fresh waters are experiencing declines in biodiversity far greater than those in the most affected terrestrial ecosystems (Strayer and Dudgeon 2010). If trends in human demands for water remain unaltered and species losses continue at current rates the probability to conserve much of the remaining biodiversity in fresh water will be very low (Figure 2).

Over 10,000 fish species live in fresh water (Lundberg et al., 2000), approximately 40% of global fish diversity and 1/4 of global vertebrate species. When amphibians, aquatic reptiles (crocodiles, turtles, etc.) and mammals (otters, river dolphins, platypus, etc.) are added to this freshwater-species total, it becomes clear that as much as 1/3 of all vertebrates are confined to fresh water.

Althought 100,000 out of approximately 1.75 million of the species described by scientists live in fresh water (Hawksworth and Kalin-Arroyo, 1995), an additional 50,000 to 100,000 species may live in ground water (Gibert and Deharveng, 2002).

Knowledge of the total diversity of fresh waters is woefully incomplete, particularly among invertebrates and microbes, and especially in tropical latitudes that support most of the world's species. Even vertebrates are incompletely known, including well-studied taxa such as fishes (Stiassny, 2002). Between 1976 and 1994 an average of 309 new fish species, approximately 1% of known fishes, were formally described or resurrected from synonymy each year (Stiassny, 1999), and this trend has continued (Lundberg et al., 2000). Among amphibians, almost 35% of 5778 species has been described during the last decade (AmphibiaWeb, 2005).

Figure 1. Biodiversity of lotic ecosystems. First line: *Equisetum sp.*, catfish and river shrimp; second line: *Natrix sp.*, Hippos and green frog; third line: cormorants, *Bufo bufo*, river crab; fourth line: *Libellula sp.*; Eurasian spoonbill and *Ephemera sp.* (credits: Roberto Cazzolla Gatti©).

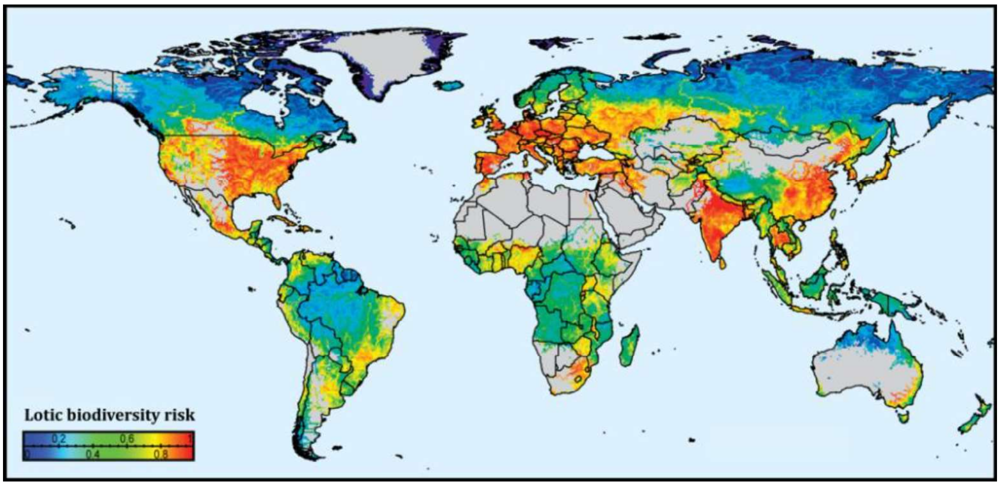

Figure 2. Biodiversity risk (from low-blue to high-red) of lotic ecosystems (adapted from Vörösmarty et al. 2010).

Adequate data on the diversity of most invertebrate groups in tropical fresh waters do not exist, but high levels of local endemism and species richness seem typical of several major groups, including mollusks, decapod crustaceans, and aquatic insects such as caddisflies and mayflies (Dudgeon, 1999, 2000; Benstead et al., 2003; Strayer et al., 2004).

Most of the prokaryote taxonomic diversity remains unexplored (Torsvik, Øvreas and Thingstad, 2002; Curtis and Sloan, 2004). It is likely that the richness of freshwater fungi and microalgae has been underestimated (Johns and Maggs, 1997; Gessner and Van Ryckegem, 2003).

In this chapter I completed an extensive review of published studies that have qualitatively and quantitatively examined the species richness and the current major threats of a special component of freshwater systems: the lotic ecosystems.

2. DIVERSITY OF LOTIC ECOSYSTEMS

Among freshwater ecosystems two different types can be recognized: lentic and lotic. This latter, from the Latin *lotus*, refers to flowing waters. Lotic waters range from springs, only a few centimeters wide, to major rivers, wide kilometers in. At the opposite, lentic ecosystems include relatively still terrestrial waters such as lakes and ponds (Marsh and Fairbridge,1999).

Moreover, lotic ecosystems can be arbitrary divided in 4 categories: large rivers, temporary/small rivers, floodplain rivers and estuarine basins (Figure 3).

2.1. Large Rivers

Large river ecosystems are the zone of the Earth with the highest biological diversity and, also, of human most intense activity (Figure 4). Rivers are important habitats for a large variety of animals and plants. Fish, amphibians, birds, insects, invertebrates, and reptiles live in rivers, or find their food there. Large rivers play a vital role in connecting habitats, and their value to plants and animals extends far beyond the surface area they cover. This habitat connectivity functions both between upstream and downstream areas, and by connecting both sides of river banks (World Rivers Review, 2011). This latter evidence pushes for an approach to management that looks at the river basin as a whole, rather than an isolated water flow. Large river biodiversity is in a state of crisis, a consequence of decades of humans exploitation with large dams, water diversions and pollution. Freshwater species are even more endangered than those on land. The pressures and impacts on the world’s large rivers have increased greatly in recent years, as a consequence of their exploitation to meet various human needs. Large rivers are particularly exposed to problems of multiple uses, often with conflicting aims. At the global scale, large river systems are altered by increased nutrient loads leading to eutrophication of river stretches, navigation and hydropower plants, which deteriorate ecosystem functions and further human uses, leading to other problems downstream.

Large dams harm biological diversity by flooding lands, fragmenting habitats, isolating species, interrupting the exchange of nutrients between ecosystems, and cutting off migration routes. They reduce water and sediment flows to downstream habitat, and change the nature of a river’s estuary, where many of the world’s fish species spawn. The impacts from dams increase the vulnerability of entire ecosystems to other threats, such as climate change (World Rivers Review, 2011).

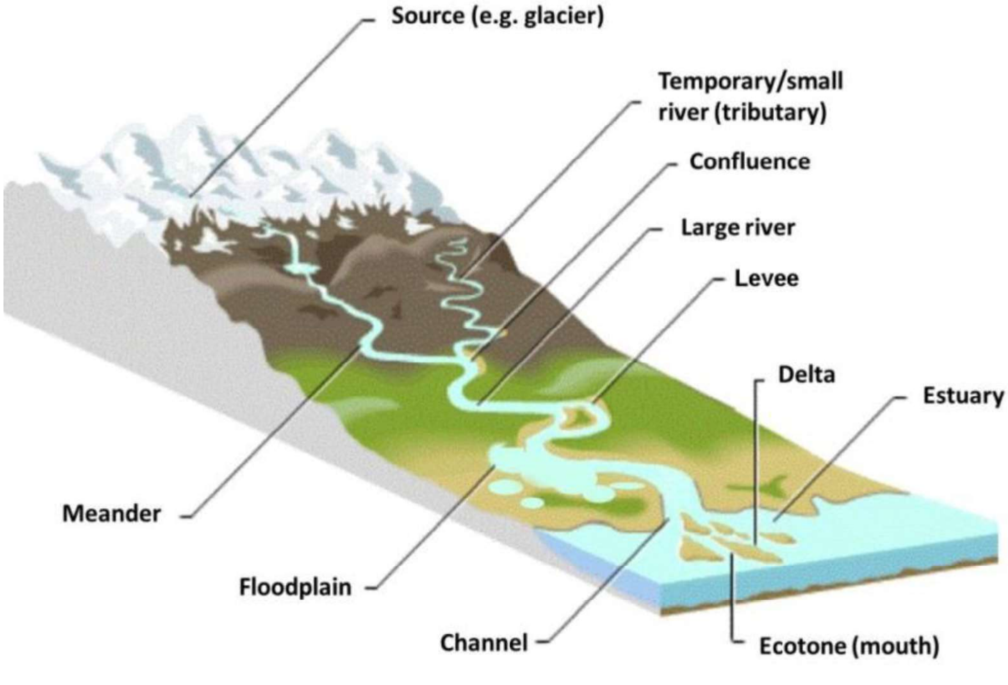

Figure 3. Diversity of the lotic systems with its different components shown.

Figure 4. An example of a large river ecosystem: the Ivindo river in Gabon (credits: Roberto Cazzolla Gatti©).

2.2. Temporary/Small Rivers

Large rivers and their riparian zones, which are considered hot spots of biodiversity (Ward et al.1999), are not the only lotic systems rich in biological diversity.

In fact, rivers that periodically cease to flow comprise a substantial proportion of the total number, length and discharge of the world's rivers (Tooth, 2000). These temporary rivers are not restricted to arid regions; they occur in most terrestrial biomes. In the next century the number and length of temporary rivers may increase in regions that experience drying trends due to climate change and to water abstraction for socio-economic uses (Larned et al., 2010). Large-scale changes in intermittence have not been considered in historical trend analyses or forecasts of future river flow patterns. However, negative trends in flow have been detected in many regions (e.g., Zhang et al., 2001; Cigizoglu, Bayazit and Onoz, 2005; Pasquini and Depetris, 2006; Milliman et al., 2008; Tockner, Uehlinger and Robinson, 2009), and interlinked climate change-runoff models predict future decreases in runoff in some mid-latitude regions (Arnell, 1999; Jones, McMahon and Bowler, 2001; Huntington, 2006; Kundzewicz et al., 2008). If these climate-driven changes will take place, increases in the occurrence and frequency of water intermittence are likely to follow.

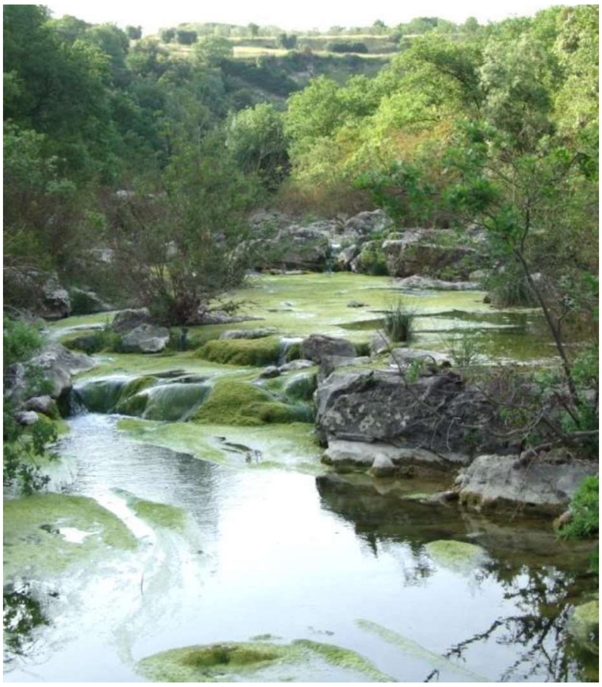

Figure 5. A temporary river formed in a South Italy canyon during the Spring (credits: Roberto Cazzolla Gatti©).

Figure 6. A small river is an important incubator of biological diversity (credits: Roberto Cazzolla Gatti©).

Temporary rivers are important links between water stored in soils, aquifers, snowpack, glaciers, vegetation and the atmosphere (Larned, 2010).

In alpine, polar and boreal catchments, meltwater from ice and snow moves to perennial rivers and lakes through networks of temporary rivers (McKnight et al., 1999; Malard, Tockne and Ward, 2000; Robinson and Matthaei, 2007). These networks expand during the melt season and contract during the freeze-up. In addition to their roles in the water cycle, temporary rivers provide a wide range of habitats.

2.3. Floodplain Rivers

Floodplain rivers are disturbance-dominated ecosystems (Figure 7) characterized by high levels of habitat diversity and biota adapted to exploit the spatial-temporal heterogeneity (Welcomme, 1979; Salo et al., 1986; Copp, 1989; Ward and Stanford, 1995; Petts and Amoros, 1996). The fluvial action of flooding and channel migration create a shifting mosaic of habitat patches across the riverine landscape. Ecotones, connectivity and succession play major roles in structuring the spatial-temporal heterogeneity leading to the high biodiversity that characterizes floodplain rivers (Ward, Tockner and Schiemer, 1999).

The most extensive and biologically diverse floodplain ecosystems in the world are found along lowland rivers of the humid tropics, including the Amazon, Orinoco and Paraguay rivers of South America, which largely retain natural flow regimes (Hamilton et al., 2007).

The complex floodplains of many large rivers offer a striking example of the influence of geomorphology on hydrology and, consequently, on ecosystem biodiversity. Much of this influence can be attributed to spatially variable patterns in the frequency and duration of soil saturation, and surface flooding (Winter, 2001; Hamilton, 2002). This variation is dictated by the elevation and position of fluvial landforms (i.e., fluvial geomorphology) in relation to the local water table and the annual range in river levels (Church, 2002). Thus, geomorphological patterns are fundamentally linked to biodiversity in floodplain environments (Brinson, 1993; Junk, 1997; Lewis et al., 2000; Ward et al., 2002).

Accumulation of water on floodplains can result from riverine overflow or from delayed drainage of local rainfall and runoff, and often these sources of water have distinct chemical and nutrient compositions (Hamilton et al. 2007). Riverine overflow often produces greater sediment and nutrient inputs and consequently higher biological productivity compared to areas flooded with locally derived waters (Klinge et al., 1990; Kalliola et al., 1991; Mertes, 1997), particularly in lowland rivers fed by mountainous watersheds. Riverine overflow tends to be episodic, albeit lasting for months in the largest rivers (Hamilton et al., 2002), while saturation, because of the emergence of local groundwater, may be constant and persist through the dry season. Soil saturation and surface flooding determine the species composition and relative abundance of plants and animals as well as the characteristics of soil, sediments, and detrital organic matter derived from the vegetation (i.e., theecological structure). In addition, saturation and flooding control biological productivity and rates of key ecological processes, such asdecomposition and biogeochemical transformations of elements (Hamilton et al. 2007). Biological activity, in turn, affects floodplain geomorphology and hydrology by influencing sediment accretion, soil development, and the flow paths of surface water and the movement of subsurface water (Figure 8). The spatial and temporal complexity of floodplain ecosystems makes them important components of regional biodiversity (Puhakka et al., 1992; Lewis et al., 2000). Floodplains, with permanent water bodies or long-lasting inundation, provide critical habitat for aquatic biota and are essential to maintain native riverine fisheries (Junk, 1997). Plants and animals on floodplains exhibit numerous adaptations to cope with, and benefit from, episodic or seasonal soil saturation or inundation.

Even relatively short episodes of saturation or inundation (from days to weeks) can profoundly affect the species composition of floodplain vegetation (Losos, 1995) and may be important in the life cycles of many aquatic animals (Junk, 1997). Variation in hydroperiod (the duration and temporal pattern of saturation or inundation), which can result from minor spatial variation in elevation, position, and soil composition of fluvial landforms, often produces dramatic differences in vegetation across floodplains and thereby contributes to biodiversity across the landscape (Salo et al., 1986; Hupp, 1988; Lamotte, 1990; Kalliola et al., 1991). Over long time scales, the migration of river channels produces landform gradients of varying age, soil development, and vegetation succession, and thereby enhances the biodiversity of environments of fluvial origin that are no longer subject to saturation or inundation (Puhakka et al., 1992).

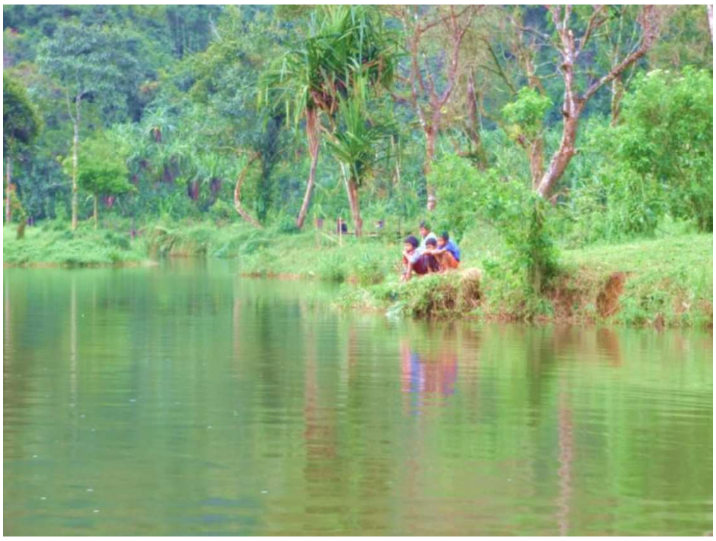

Figure 7. Floodplain rivers play a fundamental role in creating temporary habitats and are key elements for local populations' supply (credits: Roberto Cazzolla Gatti©).

Figure 8. The large foodplain ecosystem created in India by the Gange shapes the landscape (credits: Roberto Cazzolla Gatti©).

2.3. Estuarine Systems

The relatively large and unpredictable variations in salinity and water level characterizing most estuarine systems tend to select particular life-forms, limiting the number of species capable of adapting to these stresses (Rey Benayas and Scheiner 1993; Kittelson and Boyd 1997). As an ecotone between fresh and marine environments (Figure 9), estuaries and estuarine wetlands are habitats for a mixture of freshwater and halophilous plant species (Holland et al. 1990; Gosz 1993). The integrity of estuarine wetlands is maintained by hydrologic and sedimentation dynamics, environmental gradients, and lateral connections between the river or the sea and the floodplain (Junk et al. 1989; Pasternack and Brush 1998; Ward 1998). Most of the large estuarine floodplain ecosystems in the world have been altered by human activities (Bravard and Petts 1993; Vitousek et al. 1997).

Biodiversity in estuarine ecosystems is generally important because of the recurrent disturbance regime caused by the variation in salinity and water level that increase environmental heterogeneity and thus diversity (Amoros and Roux 1988; Vivian-Smith 1997). At the same time, estuarine communities are known to have a relatively low taxonomic diversity (Day et al. 1989; Costanza et al. 1993) because of the high amplitude of unpredictable stresses, which select a limited set of species adapted to changing salinity, osmotic stress, and oxygen deficiency (Chabrerie et al. 2001). Because the major factors controlling biodiversity in estuaries are easily identifiable, their wetlands are interesting open-laboratories where to study how biodiversity dynamics affect ecosystem processes (Schulze and Mooney 1994).

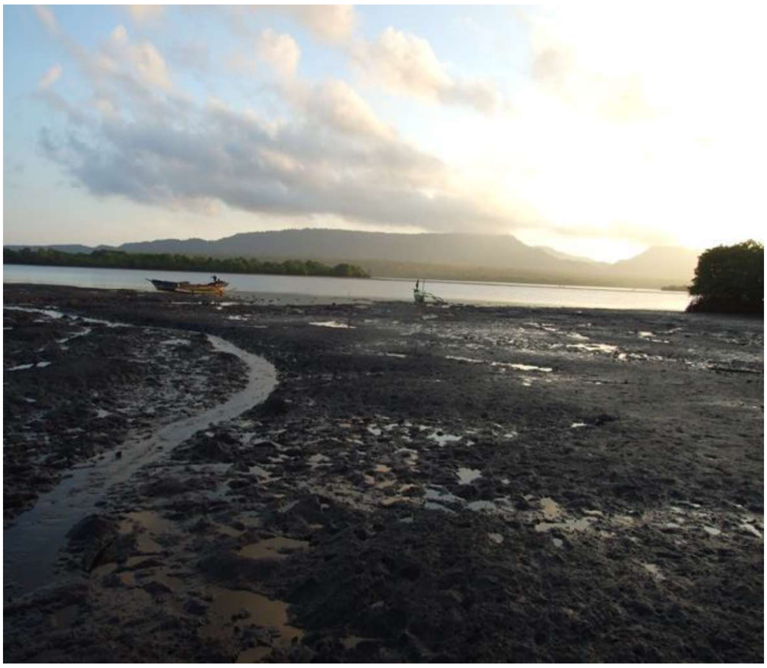

Figure 9. The esturarine ecosystems are ecotone between fresh and marine environments and habitats for a mixture of freshwater and halophilous plant species (credits: Roberto Cazzolla Gatti©).

3. SPECIAL FEATURES OF LOTIC SYSTEMS

3.1. Ecotones in the Lotic Systems

Ecotones and connectivity are interrelated structural and functional attributes of lotic ecosystems. Ecotones are transition zones between adjacent patches which, although differing from each other, exhibit high within-patch homogeneity (Ward, 1999). The ecotone concept has been a recurrent theme in ecology (Risser, 1995), with heightened interest in recent years attributable, in part, to their influence on biodiversity (Hansen and di Castri, 1992; Lachavanne and Juge, 1997). Ecotones occur over a range of scales in floodplain rivers, forming the boundaries between land and water, between surface water and ground water, and between in-stream habitat patches (Naiman and Decamps, 1990; Ward and Wiens, 1999).

In alluvial rivers, the floodplain forms a complex gradient (coarse resolution ecotone) between the river channel and the uplands, within which a variety of secondary and tertiary ecotones are embedded (Ward, 1999). These ecotones within the larger floodplains influence processes (e.g., nitrogen fixation) and structure species richness patterns (Naiman et al., 1988; Amoros et al., 1996). A variety of ecotones occur within floodplains. The littoral zone, for example, forms ecotones between the open water of floodplain lakes and the shore. These special environments also occur between different stands of floodplain vegetation (Ward, 1999). There are also vertical ecotones between surface water bodies and groundwater aquifers (Gibert et al., 1997). All of themare characterized by relatively steep gradients (e.g., thermal, chemical and organic), thereby collectively forming a high level of environmental heterogeneity across the riverine landscape.

3.2. Connectivity in the Lotic Systems

Connectivity may be defined as the ease with which organisms, matter or energy traverse the ecotones between adjacent ecological units. From a purely biological perspective, connectivity refers to gene flow between meta-populations and the extent to which ecotones alter dispersal, movement and migration (Ward, 1999).

Connectivity also refers to the extent to which nutrients, organic matter and other substances cross ecotones. Hydrological connectivity, the transfer of water between the river channel and the floodplain, and between surface and subsurface compartments, has major implications for biodiversity patterns (Welcomme, 1979; Amoros and Roux, 1988; Schiemer and Spindler, 1989; Obrdlik and Fuchs, 1991; Gibert et al., 1997; Ward, 1998). This is owing in part to the role that hydrological connectivity plays in structuring successions.

3.3. Ecological Succession in the Lotic Systems

Successional processes are responsible for much of the spatial-temporal heterogeneity of riverine floodplains (Salo et al., 1986; Amoros et al., 1987; Terborgh and Petren, 1991; Ward and Stanford, 1995; Decamps, 1996). Flooding and channel migration maintain the diversity of lotic, semi-lotic and lentic water bodies on the floodplain and create a diverse mosaic of riparian vegetation across the riverine landscape (Ward, 1999). Floodplain water bodies include the main channel and side arms (eupotamal), dead arms connected to the main channel at their downstream ends (parapotamal), abandoned braids (plesiopotamal), abandoned meander bends (palaeopotamal), alluvial spring brooks, entering tributaries, swamps and marshes (Ward, 1999). These water bodies may be arrayed along a gradient of connectivity with the main channel, from eupotamal side arms permanently connected with the main channel, to plesiopotamal habitats reconnected to the channel during the annual flood, to isolated palaeopotamal lakes that are rarely inundated by flood waters (Ward and Stanford, 1995). Therefore, each type of water body exhibits a different pattern of hydrarch succession and each is characterized by a distinctive biotic community in both surface waters (Castella et al., 1984; Copp, 1989) and ground waters (Marmonier et al., 1992).

Fluvial dynamics and channel migration also maintain a diversity of successional stages among the riparian vegetation. Salo et al. (1986) and Terborgh and Petren (1991) provide vivid descriptions of the role of natural disturbance by fluvial action in creating a mosaic of alluvial forest stands in different successional stages. Forests on the concave bends of laterally migrating rivers are undercut by erosion, and primary succession is initiated on point bars of alluvium deposited as annual increments on convex bends (Ward, 1999).

4. THREATS FOR LOTIC ECOSYSTEMS

Lotic ecosystems are increasingly impacted by multiple stressors that lead to a loss of sensitive species and an overall reduction in diversity (Meyer, 1999).

The threats to global lotic biodiversity can be grouped under seven interacting categories: overexploitation; water pollution; flow modification; destruction or degradation of habitat; invasion by exotic species; hydropower; and climate change (Arthington, 2010).

Climate change occurring at the global scale, together with nitrogen deposition, and runoff patterns (Poff, Brinson and Day, 2002, Galloway et al., 2004), are superimposed upon all of the previous threat categories.

4.1. Local Threats for Lotic Biodiversity

Overexploitation primarily affects vertebrates, mainly fishes, reptiles and some amphibians, whereas the other threat categories have consequences for all freshwater biodiversity from microbes to megafauna.

Pollution problems are pandemic, and although some industrialized countries have made considerable progress in reducing water pollution from domestic and industrial point sources, threats from excessive nutrient enrichment (Smith, 2003) and other chemicals, such as endocrine disrupters, are growing (Colburn, Dumanoski and Myers, 1996).

Table 1. Groups mostly affected by threats to lotic biodiversity. The intensity of impacts is represented by + or –

Threats to lotic biodiversity	Groups mostly affected	Intensity
Overexploitation	Plants, fishes, reptiles and some amphibians	++
Water pollution	Plants, fishes, amphibians, birds, mammals, invertebrates	+++
Flow modification	Fishes and invertebrates	++
Destruction or degradation of habitats	All groups of flora and fauna (microorganism also)	++++
Invasion by exotic species	Specialized species	+-
Hydropower	Fishes, mammals	+++

Habitat degradation is brought about by an array of interacting factors (Dudgeon et al., 2006). It may involve direct effects on the aquatic environment (such as excavation of river sand) or indirect impacts that result from changes within the drainage basin (Table 1). For example, forest clearance is usually associated with changes in surface runoff and increased river sediment loads that can lead to habitat alterations, such as shoreline erosion, smothering of littoral habitats, clogging of river bottoms, and floodplain aggradation (Dudgeon et al., 2006).

Flow modifications are ubiquitous in running waters (Dynesius and Nilsson, 1994; Vorosmarty et al., 2000; Nilsson et al., 2005). They vary in severity and type, but tend to be most aggressive in regions with highly variable flow regimes. This is because humans in these places have the greatest need for flood protection or water storage. That existing dams retain approximately 10,000 km^3 of water, the equivalent of five times the volume of all the world's rivers (Nilsson and Berggren, 2000), illustrates the global extent of human alteration of river flow. Water impoundment by dams in the Northern Hemisphere is now so great that it has caused measurable geodynamic changes in the Earth's rotation and gravitational field (Chao and Gross, 1995). Even some of the world's largest rivers now run dry for part of the year or are likely to do so as a result of large-scale water abstraction (Richter et al., 2010). Flow modifications are likely to be exacerbated by global climate change because of greater frequency of floods and droughts and, consequently, increased water-engineering responses can be anticipated (Vorosmarty et al., 2000). Impacts on river biota are likely to be severe (Dudgeon et al., 2006).

Widespread invasion and deliberate introduction of exotic species add threats to the physical and chemical impacts of humans on fresh waters, in part because exotics are most likely to successfully invade fresh waters already modified or degraded by humans (Bunn and Arthington, 2002; Koehn, 2004). There are many examples of large scale and dramatic effects of exotics on indigenous species (e.g., Nile perch, *Lates niloticus*, in Lake Victoria, the crayfish plague in Europe, salmonids in Southern Hemisphere lakes and streams; see Rahel, 2002), and impacts are projected to increase further (Sala et al., 2000).

Indirect impacts can arise from exotic terrestrial plants such as *Tamarix spp.* (Tamaricaceae), which alter the water regime of riparian soils and affect stream flows in Australia and North America (Tickner et al., 2001).

The particular vulnerability of freshwater biodiversity also reflects the fact that fresh water is a resource for humans that may be extracted, diverted, contained or contaminated in ways that compromise its value as a habitat for organisms. In some instances, impacts have been sustained over centuries and, in the case of many of the major rivers of China, they have persisted for more than 4,000 years (Dudgeon, 2000). Indeed, some authors now believe it unlikely that there remain a substantial number of water bodies that have not been irreversibly altered from their original state by human activities (Leveque and Balian, 2005).

The extent of most freshwater systems is not confined to the wetted perimeter, but includes the catchment from which water and material are drawn (Naiman and Latterell, 2005). Their position in the landscape (almost always in valley bottoms) makes lakes and rivers "receivers" of wastes, sediments and pollutants in runoff.

This is also true of seas and oceans, but inland water bodies typically lack the volume of open marine waters, limiting their capacity to dilute contaminants or mitigate other impacts (Dudgeon, 2006).

In addition, in many parts of the world fresh water is subject to severe competition among multiple human stakeholders, to the point that armed conflicts can arise when water supplies are limiting and rivers traverse political boundaries (Poff et al., 2010). There are 263 international rivers, draining 45% of the Earth's land surface, and this area supports more than 40% of the global human population (Postel and Richter, 2003). In the vast majority of disagreements over multiple uses of water, whether they are international or on a local scale, allocation of water to maintain aquatic biodiversity is largely disregarded (Poff et al., 2003). In China and India, where approximately 55% of the world's large dams are situated, hardlyany consideration has been given to the downstream allocation of water for biodiversity (Tharme, 2003; Poff et al., 2010).

The combined and interacting influences of the major threat categories have resulted in population declines and range reduction of freshwater biodiversity worldwide.

Qualitative data suggest reductions in numerous wetland and water margin vertebrates (e.g., a loss of 19 mammals, 92 birds, 72 reptiles and 44 fish species), while population trends indicate declines averaging 54% among freshwater vertebrates (mainly waterfowl), with a tendency toward higher values in tropical latitudes (Groombridge and Jenkins, 2000; Loh, 2000). Furthermore, 32% of the world's amphibian species now are threatened with extinction, a much higher proportion than threatened birds (12%) or mammals (23%), and 168 species may already be extinct (AmphibiaWeb, 2005). The well-known global decline of amphibians started during the 1950s and 1960s and has continued at the current rate of approximately 2% per year, with more pronounced decreases in tropical streams (Houlahan et al., 2000; Stuart et al., 2004). This is close to the estimate of 2.4% for declines in populations of freshwater vertebrates over the period 1970–1999 (Balmford et al., 2002). These estimates are extremely alarming. The limited data on extinction rates from few continent are believed to be indicative of a global freshwater "biodiversity crisis" (Kottelat and Whitten, 1996).

Rates of species loss from fresh waters in non-temperate latitudes are not known with any degree of certainty. They are likely to be high because species richness of many freshwater taxa (e.g., fishes, macrophytes, decapod crustaceans) increases toward the tropics. The drainage basins of many large tropical and subtropical rivers (e.g., the Ganges and Yangtze) are densely populated, with large dams, altered flow patterns and gross pollution from a variety of sources being the inevitable outcomes (Dudgeon, 2000, 2002). For larger species in these rivers, the situation is parlous: the Yangtze dolphin (*Lipotes vexillifer*) is probably the most endangered mammal on Earth (now numbering fewer than 100 individuals; Dudgeon, 2005), and the Ganges dolphin (*Platanista gangetica*) is “Endangered” (IUCN, 2003). The crocodilian fauna of the Ganges and Yangtze likewise consists entirely of threatened or highly endangered species. Many other species of water associated reptiles,a primarily tropical group, are gravely threatened (Gibbons et al., 2000), most particularly turtles, as are large freshwater fishes in most rivers (Hogan et al., 2004), and many freshwater fish stocks are over-fished to the point of population collapse (Dudgeon, 2002).

Floodplain ecosystems tend to be strongly impacted by the impoundment of river channels and other modifications that alter and generally reduce the natural hydrological variability of parent rivers. These floodplains lie along rivers with very low longitudinal slopes. Major river alterations for purposes such as navigation and hydroelectric generation have recently been proposed for each of these river systems, and such projects could affect extensive areas of floodplains through alteration of the natural flood regime (Hamilton, 2002).

Hydropower, which has substantial unexploited potential in many developing countries, can potentially mitigate greenhouse gas emissions by displacing fossil fuel production of energy, but large scale hydropower systems, in particular, can have adverse biodiversity and social effects (Secretariat of the Convention on Biological Diversity, 2009).

The environmental and social impacts of hydropower projects vary widely (Table 2), depending upon pre-dam conditions, maintenance of upstream water flows and ecosystem integrity, design and management of the dam (e.g., water-flow management) and the area, depth and length of the reservoir. Run of the river dams typically have fewer adverse environmental and social effects. Sectorial environmental assessments can assist in designing systems with minimum adverse consequences for ecological systems (Secretariat of the Convention on Biological Diversity, 2009).

Table 2. Environmental impacts of hydropower stations and operations (adapted from McCully, 1996)

Impacts due to Hydropower presence	**Impacts due to Hydropower operation**
Upstream change from river valley to reservoir Flooding of terrestrial habitats Conversion of terrestrial habitats Changes in fish migration patterns Changes in vegetation	Changes in hydrology Changes in downstream and total flows Changes in seasonal and short term timing and fluctuation in flows Changes in high and low flows Local extinctions of fish species Changes in vegetation
Changes in downstream morphology of banks, delta, riverbeds, estuary and coastline due to altered sediment load	Changes in downstream morphology due to altered flow patterns Reduction of connectivity
Changes in water quality, river temperature, nutrient load, turbidity, dissolved organic material, heavy metals and minerals	Changes in downstream water quality due to altered flow patterns Changes in microorganism temporary communities
Loss of biological diversity Loss of animal migrations Changes in ecological succession	Decrease of riverine, riparian and floodplain habitat diversity Elimination of floods

Well-designed installations, for example using modern technologies that cascade the water through a number of smaller dams and power plants, may reduce the adverse environmental impacts of the system (Secretariat of the Convention on Biological Diversity, 2009).

Small and micro-scale hydroelectric schemes normally have low environmental impacts, but the cumulative effects of many projects within a watercourse may have considerable impact on the biodiversity within a larger area. In general, run-of-river projects will have fewer impacts than storage dams with large reservoirs but they may also have serious effects on biodiversity.

These impacts are mainly due to the blocking of fish migration, either because of the physical barrier of the dam wall or through the dewatering of a stretch of river below the dam (Table 2).

Another important determinant of the impacts of dams is their location within the river system. Dams near the headwaters of tributaries will tend to have fewer impacts than mainstream dams that may cause perturbations throughout the whole watershed (Pringle 1997). The protection of dams from siltation may be a mayor incentive for biodiversity conservation in the form of reforestation or afforestation measures within the watershed. The World Commission on Dams has published a comprehensive list of guidelines for water and energy planning which might be helpful in that respect (World Commission on Dams 2000).

4.2. Global Threats to Lotic Biodiversity

Global biodiversity hotspots contain exceptional concentrations of endemic species in areas of escalating habitat loss. However, most hotspots are geographically constrained and consequently vulnerable to climate change (Table 3) as there is limited ability for the movement of species to less hostile conditions. Predicted changes to rainfall and temperature will undoubtedly further impact on freshwater ecosystems in these hotspots (Davies, 2010). For instance, current and predicted water temperatures may exceed thermal tolerances of aquatic fauna (Pörtner and Knust, 2007). Consequently, fauna cannot change their distribution southwards or with altitude as a response to increasing temperatures. Therefore, any mitigation responses need to be in situ to produce a suitable biophysical envelope to enhance species' resilience (Davies, 2010). This could be done through "over restoration" by increased riparian replanting at a catchment scale.

According to IPCC (2007), inland aquatic ecosystems are highly vulnerable to climate change, especially in Africa. Higher temperatures will cause water quality to deteriorate and will have negative impacts on microorganisms and benthic invertebrates. Plankton communities and their associated food-webs are likely to change in composition. Distributions of fish and other aquatic organisms are likely to shift northwards and some extinctions are likely. Changes in hydrology and abiotic processes induced by changes in precipitation, as well as other anthropogenic pressures, will have large impacts on aquatic ecosystems. Boreal peatlands will be affected most and suffer major changes in species composition. Many lakes will dry out. Increases in the variability of precipitation regimes will also have important impacts and may cause biodiversity loss in some wetlands. Seasonal migration patterns of wetland species will be disrupted (Secretariat of the Convention on Biological Diversity, 2009). The impacts of increased CO_2 will differ among wetland types, but may increase NPP in some systems and stimulate methane production in others. On the whole, ecosystem goods and services from aquatic systems are expected to deteriorate.

Climate change is expected to impact lotic ecosystems in two major ways: first, through changes in the water cycle; second, through associated changes in the terrestrial ecosystems within a given catchment (Table 3).

For inland wetlands, changes in rainfall and flooding patterns, across large areas of arid land, will adversely affect bird species that rely on a network of wetlands and lakes that are alternately, or even episodically, wet and fresh, and drier and saline (Roshier and Rumbachs, 2004).

Responses to these climate induced changes may also be affected by fragmentation of habitats or disruption or loss of migration corridors or, even, changes to other biota, such as increased exposure to predators by wading birds (Butler and Vennesland, 2000).

The lack of thermal refugia and migratory routes in streams and rivers may cause contraction of the distributions of many fish species. For example, warmer lake water temperature will reduce dissolved oxygen concentration and lower the level of the thermocline, most likely resulting in a loss of habitat for cold-water fish species. In addition, reduced summer flows and increased temperatures will cause a loss of suitable habitat for cool water fish species in riverine environments (Bunn and Arthington, 2002).

For streams, the effects of temperature-dependent changes would be least in the tropics, moderate at mid-latitudes, and pronounced in high latitudes where the largest changes in temperature are projected. Increased temperatures will alter thermal cycles of lakes and solubility of oxygen and other materials, and thus affect ecosystem structure and function (Secretariat of the Convention on Biological Diversity, 2009). Changes in rainfall frequency and intensity, combined with land-use change in watershed areas, has led to increased soil erosion and siltation in rivers. This, along with increased use of manure, chemical fertilizers, pesticides, and herbicides as well as atmospheric nitrogen deposition, affects river chemistry and has led to eutrophication, with major implications for water quality, species composition, and fisheries. The extent and the duration of the ice cover is projected to decrease in some high latitude lakes and, thus, the biodiversity may be affected by the shorter ice cover season (Christensen et al., 2003).

Climate change will have most pronounced effects on wetlands through altering the hydrological regime as most inland wetland processes are intricately dependent on the hydrology of the catchments (river basin) or coastal waters. This is expected to affect biodiversity and the phenology of wetland species (van Dam et al. 2002)

As with terrestrial ecosystems, adaptation strategies to climate change in lotic ecosystems include conservation and spatial linkages (Secretariat of the Convention on Biological Diversity, 2009).

Adaptation options to these changes should consider all components of the watershed (Sparks 1995). River biota, within reasonable limits, is naturally well adapted to rapid and unpredictable changes in environmental conditions (Puckridge et al. 1998). For rivers, it may be essential to conserve or restore ecosystem connectivity, both longitudinally along the river course and laterally between the river and its wetlands, in order to sustain ecosystem function (Ward et al. 2001). However, many of the natural aquatic corridors are already blocked through dams and embankments. This increases the vulnerability of lotic biodiversity to climate change and constrains implementing adaptive strategies. In their lower reaches, coastal rivers enter the estuarine and coastal zone where they have a major influence. These areas should be considered a contiguous part of inland water ecosystems and managed together under the ecosystem approach. The identification of the degree of vulnerability of the various components of complex inland water ecosystems, and the subsequent development of appropriate ecosystem management plans based upon this information, is a critical requirement for adaptation to climate change for inland waters (Andrade, Herrera, and Cazzolla Gatti, 2010).

Any management that favours near natural hydrological function in inland water ecosystems is likely to have major benefits for the conservation of biodiversity. In particular, modern approaches to the management of rivers recognise that for many systems change is inevitable. This has stimulated much interest in the concept of sustaining "environmental flows" as a management target for rivers (Tharme, 2003). Such approaches need to take on board climate change if they are to be adaptive. The increase in extreme weather events that climate change may bring (for freshwaters particularly the frequency and extent of droughts and floods) is likely to be more of a concern with isolated lakes and wetlands. The issue of extreme hydrological events is, however, of major significance to integrated water resources planning and management. For example, maintaining river floodplains and wetlands helps restore water balance and hence mitigate catastrophic flooding (Secretariat of the Convention on Biological Diversity, 2009).

Table 3. Impacts of climate change on lotic biodiversity and potential adaptation strategies

Impacts of climate change	**Involved organisms**	**Potential adaptations**	**Extinction risk level**
Water temperature increase	Fishes, plants, crustaceans, benthic species, turtles and mammals	Migrations northwards and upwards when not limited	Very High
Precipitation patterns changes	Fishes, vegetation, benthic invertebrates	Dispersal and in situ adaptations	Moderate
Water quality decline due to higher temperature	Microorganisms, benthic invertebrates, plankton communities	Dispersal and in situ adaptations	High
Fragmentation of habitats, disruption or loss of migration corridors	Fishes, reptiles and mammals	Behavioural changes and phenotypic plasticity for generalist species	High
Invasion of species	Specialized species	Avoidance of competition (shifting of niches)	Low-moderate
Extreme events	All species (benthic species less affected)	Long distance dispersal	Moderate
Community composition changes	All species	New interactions and mutualism partnerships	High

Climate change, therefore, can be regarded as providing additional incentives to manage inland waters better and both the financial and conservation benefits of doing so are considerable. Maintaining natural river form and related ecosystem processes is likely to provide significant benefits for coastal regions and populations. For instance, while maintaining their traditional way of life, lotic species also play a significant role in nutrition of the nomadic reindeer-herders who take long stops at rivers and lakes for fishing. Hunting is the oldest occupation of mankind, and many reindeer hunters migrated across vast distances of Siberia throughout historic times (Secretariat of the Convention on Biological Diversity, 2009).

Major impacts of climate change that need to be addressed in water management include increasing flood risk, increasing risk of drought, and change in timing of flow regimes. Common technical approaches to flood risk include the construction of dykes and dams. Technical solutions are also often applied to address problems of water shortage, including the construction of reservoirs and canals, facilities for water diversion and abstraction from rivers, and alterations to river beds to improve shipping capacity during low-water periods. Hard structures can have significant environmental impacts, such as destruction or alteration of wetlands, reducing connectivity between lakes, rivers and riparian zones, and changing sediment flows (Secretariat of the Convention on Biological Diversity, 2009). Restoration of upland watersheds and floodplain restoration are ecologically viable alternatives that deserve attention.

Climate change is leading to increased inland flooding in many regions through more variable rainfall events. Restoring and maintaining ecosystems in upland watersheds, including through the management of soils and vegetation, can contribute to reduce the risk of flooding and maintain regular water supplies. Run-off from mountainous areas in small islands is often the major supply of water, and in many countries, watersheds form a critical part of the national economy (Secretariat of the Convention on Biological Diversity, 2009). Often these watersheds are degraded, and their rehabilitation is one adaptation option.

Wetland ecosystems in watersheds can reduce flooding and sediment deposition whilst improving water quality downstream. A study of upland forests in a watershed in Madagascar (Kramer et al., 1997) has estimated their flood protection value at $126,700, and peat bogs in Sri Lanka that buffer floodwaters from rivers have an estimated annual value of more than $5 million (Emerton and Kekulandala, 2003). In the Morogoro region of the United Republic of Tanzania, reduced river flow and increased flooding has been attributed to deforestation in the mountains, and it has been suggested that effective management of soil, forests and water resources are needed as adaptation measures, along with improved social capacity (Chamshama and Nduwayezu, 2002). Ecuador and Argentina have integrated forests and wetlands into their living with floods strategies, and reforestation is recognised as an important option for adaptation in the watersheds of the Philippines. Vietnam includes measures such as integrated management of watersheds in its disaster reduction planning, along with forest management, and soil and water conservation. Large-scale afforestation projects in China have been carried out with the aim of reducing flooding and increasing water conservation, and countries of Central America are collaborating to protect watersheds and forests (Campbell et al., 2009).

Climate change is causing an increase in the scale of flooding and dry periods in many flood plains. In some systems dams are no longer a viable adaptation strategy, and in some cases dams have had negative environmental and socio-economic impacts. Also in these circumstances ecosystem management is an effective adaptation strategy at the river basin scale and an alternative to the development of small-scale dams. In developed countries, cost-effective flood reduction strategies, which allow re-growth of vegetation alongside rivers and establish vegetation buffers along streams, combined with the reduced development of infrastructure, are being promoted in some areas. Restoration of floodplain ecosystems can also help to reduce the levels of water pollution following extreme events (CBD, 2009).

There is new observational evidence of climatic impacts on inland waters. A recent research has shown that there has been a fourfold increase in permanently dry ponds in Yellowstone over the last 16 years and that this can be linked directly to dramatic declines in amphibian populations and diversity (McMenamin et al. 2008). Modelling work suggests that climatic warming, in combination with other environmental changes, may cause the nature of river channels to change in the Russian Arctic (Anisimov and Reneva, 2006). In the Arctic, factors such as reduced ice-cover duration on lakes, especially in northern Arctic areas, increased rates of rapid stratification and primary production, and decreased oxygenation at depth. This will possibly result in a reduction in the quality and quantity of habitat for species such as lake trout. Decreased water flow in summer is also likely to decrease habitat availability and possibly deny or shift access for migrating fish (Reist et al., 2006; Wrona et al., 2006). In monsoonal Asia, where ecological processes surrounding rivers are mediated by flow, disruptions in timing and velocity will have large environmental impacts (Dudgeon, 2011). The interaction between climate change and land cover change is likely to lead to reduced discharge from many rivers that will in turn lead to significant loss of freshwater fish species (Xenopoulos and Lodge, 2006).

There is also new observational evidence of compositional change in fish communities in France over the last 15-25 years (Daufresne and Boet, 2007): species richness, proportions of warm water species, and total abundance increased. Modelling has demonstrated negative impacts on the habitats of native fish species, including freshwater salmon (Xenopoulos and Lodge, 2006; Battin et al., 2007), especially at higher elevations and in headwater areas (Buisson, 2008). In the Arctic, there is an expected decrease of native fish as southern Arctic and sub-Arctic fish species migrate northwards. The broad whitefish (Arctic char complex), and the Arctic cisco are particularly vulnerable to displacement. Decreased water flow in summer is likely to decrease habitat availability and possibly deny or shift access for migrating fish (Reist et al. 2006; Wrona et al.; Anisimov et al. 2007; Berry 2008). Temperature is a very important determinant of distribution and survival of aquatic macro-invertebrates at high latitudes, and changes in species composition have already been shown for boreal inland waters (Heino et al. 2009).

It has been suggested that species characteristic of lentic systems may disperse more effectively than those of lotic systems (Hof et al. 2008), and therefore that lentic systems may show more rapid compositional change in response to changing climate (Heino et al. 2009).

Models show that climate change will also affect wetland species composition through its effects on river flow, especially low water flows (Xenopoulos and Lodge 2006; Harrison et al. 2008), though the interaction with socio-economic drivers of flow management is also very important.

There is considerable and growing concern about the linkages between climate change impacts on aquatic systems (including warmer water temperatures, shorter duration of ice cover, altered streamflow patterns, increased salinization, and increased demand for water storage and conveyance structures) and aquatic invasive species (Rahel and Olden 2008).

Climate change is influencing invasive establishment by eliminating adverse winter conditions and will alter the distribution and ecological impacts of existing invasive species, by enhancing their competitive and predatory effects on native species, and by increasing the virulence of some diseases (Hellmann et al., 2008; Rahel and Olden, 2008).

Predictions done for Canada indicate that water temperature may change as much as 18º C by 2100, which would mean that a number of lakes will be newly vulnerable to invasion by smallmouth bass (Sharma et al., 2007).

Other factors that will interact with climate change in determining compositional change in inland waters include acidification (Barnes and Conlan, 2007; Durance and Ormerod, 2007), eutrophication (Heino et al. 2009) and land cover change, with change in composition of terrestrial systems (Chapin et al., 2005; Heino et al., 2009) and agricultural expansion (Heino et al., 2009).

There is very little information on real or projected changes in aquatic ecosystems in the tropics, but it is clear that some major tropical wetlands are at risk from altered flows of freshwater (Gopal and Chauhan, 2006; Xenopoulos et al., 2005; Xenopoulos and Lodge, 2006).

Some others significant impacts of climate change have been projected for both carbon storage and fisheries services from inland waters. While this has been projected particularly strongly for the Arctic (Wrona et al., 2006), there is also a growing body of model-based evidence relating to other regions (Xenopoulos and Lodge, 2006). A new concern is the effect of sea level rise on carbon storage in coastal wetlands, including 150,000 km^2 of freshwater peatlands worldwide below 5 m elevation and vulnerable to sea level rise, which are likely to emit significant amounts of carbon when they are inundated (Henman and Poulter, 2008). The protection and biodiversity conservation roles of coastal wetlands are also at risk as, for example, in the case of the Sundarbans, the world's largest wetland, which is threatened by altered freshwater flows and sea level rise, which are both influenced by climate change (Gopal and Chauhan, 2006). Ecosystem services provided by peatlands are also at risk, as temperature changes are expected to reduce their function as carbon sinks (Lloyd, 2008).

4.3. Global Scenarios for Lotic Biodiversity

Scenarios and projection of recent trends suggest that a combination of climate change, water withdrawal, pollution, invasive species, and dam construction will further deteriorate the current state of freshwater biodiversity (Leadley, 2010). The particular vulnerability of freshwater species to global changes reflects the fact that both fish and freshwater are resources that have been heavily managed.

Scenarios for freshwater biodiversity are limited compared to terrestrial and marine biodiversity (Sala et al., 2000). Moreover, global scenarios tend to address water resources for people, but rarely include models of freshwater biodiversity. Those that do it, model a limited number of drivers and lack or treat only qualitatively major drivers, such as dam construction, eutrophication and invasive species.

Habitat loss and/or fragmentation are among the greatest threats to biodiversity worldwide, and this certainly holds true for riverine fish (Dudgeon et al., 2006). It is almost certain that disturbances to freshwater ecosystems, such as dams, reservoirs and diversions for irrigation and industry, will endanger or extinguish many freshwater fish species in the future, by creating physical barriers to normal movements and migration of the biota and by decreasing habitat availability.

Currently it is difficult to make precise predictions about how climate change will affect fish biodiversity, even though climate niche modelling suggests that, locally, the number of warm-water species may increase in temperate areas and some cryophilic (i.e., cold-water) species may regionally vanish (Leadley, 2010).

Narrow endemic riverine fishes can be particularly threatened by climate change. The biggest problems occur in basins which have an East-West configuration, while in basins with a North-South configuration, there will be more opportunities for migration and adaptation, as long as the rivers are not blocked by dams (Leadly, 2010). Models also project that in shallow lakes in northern latitudes there will be summer fish kills of cold-water species due to both increased water temperatures and decreased dissolved oxygen.

Other negative impacts of climate change on freshwater ecosystems are changes in snow melt timing and flow volumes. Global climate scenarios have been applied to known relationships between fish diversity and river discharge. Results predict decreased freshwater biodiversity in about 15% of the world’s rivers in 2100, from a combination of reduced run-off (caused by climate change) and increased water withdrawals for human use. However, these predictions should be considered with great caution, as the approach does not provide true extinction rates, but instead a percentage of species “committed to extinction” with an unspecified time lag. These predictions also do not include other current stresses on freshwater fish, such as pollution or river fragmentation (Olden et al., 2010).

Based on the established relationship between the number of non-native fish species and human activity, we expect that river basins of developing countries will host an increasing number of non-native fish species as a direct result of economic development (Leprieur, 2008).

Furthermore impoundments and climate change may facilitate the expansion of invasive species and diseases associated with lake ecosystems (Johnson et al., 2008).

Pressure on freshwater ecosystem services (Rapport et al., 1998) and wetland degradation will increase leading to the deterioration of regulating services such as regulation of water quality and flood protection.

The combination of population growth, increasing water use and climate change will lead to an increase in human population living in river basins facing severe water stress. This will not only increase the risks of chronic water shortages in these regions, but will also cause major negative impacts on freshwater ecosystems (Leadley, 2010).

Many studies report that eutrophication of freshwater systems will increase in the developing world as fertilizer use and untreated sewage effluents continue to increase (Tilman et al., 2001). This may be further exacerbated in some regions by decreasing precipitation and increasing water stress. The transition to eutrophic conditions is in some instances difficult to reverse and can lead to loss of fish species, loss of recreational value, and in certain cases health risks for humans and livestock (Leadley, 2010).

4.4. Remote Sensing to Monitor Biodiversity of Lotic Ecosystems

Most small rivers are not directly detectable using data at, or below the, 30 m resolution threshold, considered a practical limit for national assessments. Often, analysts turn to watershed analysis within geographic information systems (GIS) software (Gardiner and Díaz-Delgado, 2007). These methods use digital elevation model (DEM) data to infer the direction that water would flow if travelling overland and downhill from any point in a landscape (O'Callaghan and Mark, 1984). Once flow direction information has been extracted, it is possible to infer the total number of cells, and hence area, above every cell in a DEM. The GIS database could include watershed and river attributes such as forested area, land use, river discharge at measured locations, resident species, important features such as dams, water withdrawal points, discharge points (Figure 10), or other information pertinent to hydrology or biodiversity of wetlands (Hutchinson, 1991). These GIS methods provide a means for tracking physical and biological conditions in rivers and watersheds.

For instance, HydroSHEDS (Hydrological data and maps based on SHuttle Elevation Derivatives at multiple Scales) represents waterbodies, waterways, watersheds, and surface hydrology on a near-global basis and at multiple resolutions (Lehner et al. 2006). The data were built from NASA's SRTM data, which describe surface elevations for Earth's land area lying between ±57 degrees latitude. HydroSHEDS data may be downloaded free of charge (http://hydrosheds.cr.usgs.gov/). The goal of developing this database was to generate key data layers to support watershed analyses, hydrologic modelling, and freshwater conservation planning at previously inaccessible quality, resolution, and extent. The seamless coverage of HydroSHEDS makes this dataset useful for continental analyses because it eliminates the need to blend multiple data sources (Gardiner and Díaz-Delgado, 2007).

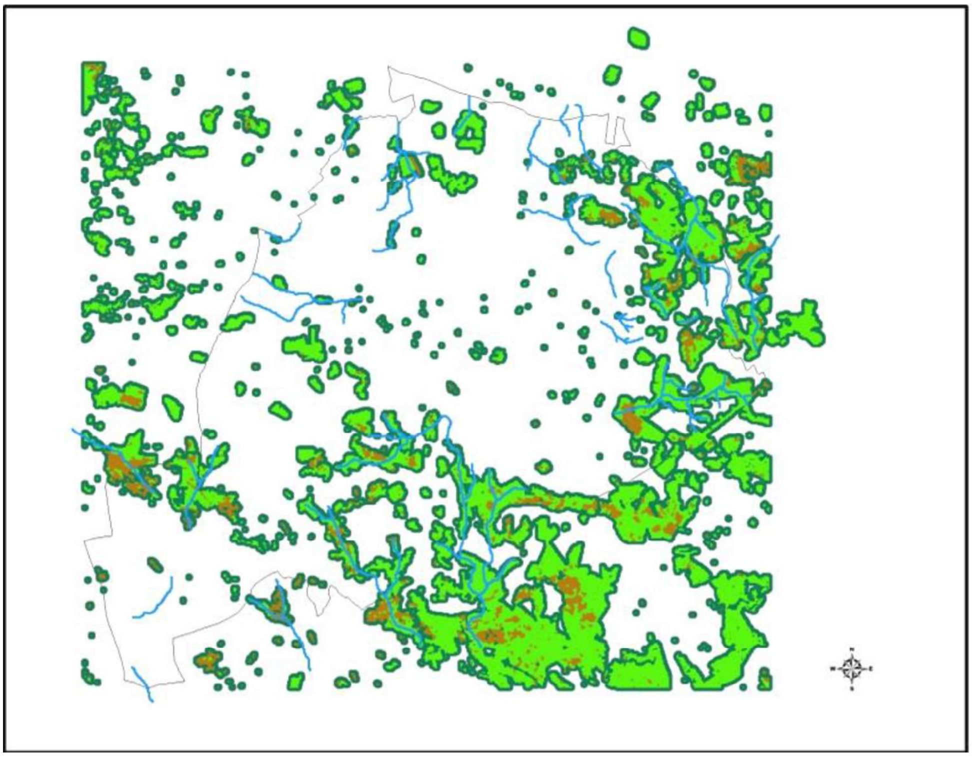

Figure 10. An elaboration derived from satellite maps of a region in South Italy that shows the hydrographic network and vegetation cover (green: woodlands; brown: grasslands). This map has been used by WWF to address conservation policies of the region (from Cazzolla Gatti, 2009).

Data acquired at high- and low-water are needed to accurately map wetland extent because these ecosystems are defined by water level fluctuation and function differently at different water levels. The acquisition time of archived data may be compared to the best-known stage information for a given area in order to choose imagery that will be of most use for characterizing high- and low water regimes of the targeted ecosystem. For upland wetlands, antecedent soil moisture and precipitation affect river discharge, wetland extent, and surface water levels and therefore influence the optimal date for acquiring imagery. The situation is more complex for seasonally inundated systems that receive water input from rivers because river flood stage and peak flow are influenced by upstream, catchment-wide hydrological factors that delay peak discharge relative to peak precipitation events (Gardiner and Díaz-Delgado, 2007). Temperate rivers reach bank full stage with a recurrence interval of between 1.5 and 2.5 years, suggesting that optimal data for analysis might appropriately come from different years in order to capture both low- and high-water events within the data record for a site. Mapping the high- and low-water stages of rivers with active floodplain systems, such as the Amazon, is important since the function of these systems is defined relative to that variation in water level and extent.

The behaviour, function, and aerial extent of wetlands changes through time, so characterizing changes in habitat quality requires a thorough understanding of natural variability. Water and nutrients are transported from headwaters to downstream river ecosystems, but there is a bidirectional interaction of rivers with active floodplains (Gardiner and Díaz-Delgado, 2007). Nutrient flux from floodplains into river food webs is an important linkage between rivers and the floodplain habitat associated with them. When researchers first mapped and compared the extent of floodplain inundation at high and low-water levels, they surmised that the mass of carbon emitted from rivers in the form of CO_2 during high water stages throughout the year was comparable in magnitude to the amount of carbon transported down river (see Melack, 2004). Thus, accurate mapping at high and low-water levels has led to significant new insights into the structure and function of aquatic systems.

Recent remote sensing efforts have demonstrated how to map floodplain forests using a variety of satellite sensors and data available at multiple resolutions. Hamilton and colleagues (2007) used remotely sensed data in combination with HydroSHEDS river network data to characterize wetlands in floodplains of the Madre de Dios River. They mapped floodplains, standing water, and vegetation associated with unique geomorphic settings in this flood-dominated ecosystem. This research employed object-oriented, contextual classification, a set of techniques that utilizes the spatial setting of landscape features to help identify and classify imagery. Image data included Landsat 7 ETM+ data, elevation profiles from NASA’s SRTM, and JERS-1 L-band radar scatterometer mosaics.

To evaluate in situ biological and physical properties of river ecosystems, researchers often use “multi-metric” indicators, statistical descriptions that simultaneously describe a site’s species and local habitat relative to undisturbed sites with similar landscape settings (Gardiner and Díaz-Delgado, 2007). Multi-metric indicators are derived empirically from a set of sites or through time, for example from density or relative proportion of taxa collected at a site or group of sites. The categories and point assignments used to derive multi-metric scoring systems must be calibrated to the fishes, macroinvertebrates, or microbes found in streams and rivers within a bioregion, so this work is conducted by a biologist with the requisite expertise in regional fauna and flora. Fish and invertebrate ecologists have the most experience using multi-metrics to describe and categorize ecosystem health in rivers and streams, but taxonomists and ecologists are studying how to develop indicators of stream health that focus on the microbes found at a stream sampling site. Multi-metric scores can be compared statistically to land use data derived from remote sensing and extracted on a watershed basis using GIS software (Gardiner and Díaz-Delgado, 2007). This statistical approach guides inferences about the effect of watershed practices on streams or rivers. This procedure is widely practiced, but should be conducted only through direct collaboration among experts in GIS, remote sensing, and freshwater biology.

Hyperspectral technologies have also been used to study primary productivity of inland waters (Hoogenboom et al., 1998), although these studies focus on very small areas and use data not available on a global basis. Optical data are also used to estimate suspended solids concentrations in large water bodies (Dekker et al., 2001).

Change and variability are inherent to the structure and functioning of wetlands. Just as one may assess the natural variability of water extent, exogenous inputs, and biota within wetlands, it is possible to measure long-term trends and changes to wetlands using the same or similar methods. Some changes to wetlands can be evaluated somewhat directly, for example the influence of land cover change on the timing and delivery of water and suspended constituents to rivers, or the effect of global warming on boreal wetlands (Gardiner and Díaz-Delgado, 2007).

Land cover change upstream of receiving waters alters the hydrologic, nutrient, and physical templates of those ecosystems. When forested catchments are clear-cut in temperate forest ecosystems, recovery of some parameters, such as nutrient retention and turnover, requires up to several years to re-establish pre-disturbance regimes. Other physical characteristics require decades for recovery. For example, sediment delivered to rivers and streams following a major disturbance, such as watershed-wide clear-cutting, may require infrequent, episodic torrential rain events in order to generate sufficient hydrologic power to redistribute large quantities of sediment downstream. Once vegetation recovers, the legacy of historic deforestation events can therefore have a very long-lasting impact on the habitat template of stream ecosystems (Gardiner and Díaz-Delgado, 2007).

Spatial data describing regional climate patterns, physiography, land cover, and land use lend insight into how to manage watersheds. Conservation planners prioritize their effort using the best available data describing a region of interest. Often, data describing biodiversity are absent, so planning must move forward in the absence of biological information using surrogate measures, such as climate information (Gardiner and Díaz-Delgado, 2007).

Planners and researchers from WWF-US, Michigan State University, Woods Hole Research Center, and WWF-Peru prioritized conservation recommendations for a 160,000 km2 headwater region of the Madre de Dios and Orthon rivers in Peru (Thieme et al. 2007). Each river is a tributary of the Amazon River, and the study area as a whole is within the south-western Amazonian Moist Forests ecoregion of the Global 200 priority regions identified by Olsen and Dinerstein (1998). The work used GIS-based analyses of terrain, vegetation, and existing protected areas to recommend areas for conservation attention. The study's authors hope the work will prevent problems arising from road building and other land-clearing activities that are likely to accompany oil and gas exploration in the region. Activities like these will remove vegetation and expose soil, thereby increasing sediment delivery to waterways through erosion and transport of disturbed soil. Sedimentation is among the most common processes that degrade river ecosystems (Beechie et al., 2010). GIS data describing watershed boundaries, stream channels, and watershed morphometry provided requisite data for evaluating potential discharge along stream segments, percentage of watershed area found within the Andes, and connectedness of river segments among protected areas that have already been identified. Watershed-based analyses, such as those conducted in the Madre de Dios River basin, are an essential component of evaluating the potential influence of land use decisions on wetlands (Gardiner and Díaz-Delgado, 2007).

Remote sensing analyses complement campaigns focused on the structure and function of wetland ecosystems. For example, due to changes in freeze-thaw cycles and permafrost conditions stemming from global warming, there is increasing attention and interest in greenhouse gas emissions from boreal forests (Turner et al., 2009).

Approximately 25% of the carbon that is bound within terrestrial ecosystems is likely found in high-latitude peat lands (Hess and Melack 1994). When they dry out, peat lands respire CO_2 and CH_4 into the atmosphere, so monitoring inundation in these areas is important for quantifying greenhouse gas emissions from peat lands. In boreal Siberia and eastern Canada, Gorham (1991) argued that satellite data may be used to monitor the declining area of open water as an indicator of global warming effects on peat land ecosystems, but that effort also requires contribution from biogeochemists with expertise in quantifying and evaluating outgassing. Conversely, biogeochemists recently estimated that methane emissions from seasonally melted lakes in permafrost regions of Siberia may contribute about twice as much CH_4 to the atmosphere as previously thought (Walter et al. 2006). Their estimates were based on field-collected samples from a handful of lakes, and results were extrapolated based on estimates using GIS and remote sensing. Kimball and colleagues (2006) have used microwave data to show that seasonality has changed in recent decades, with warm temperatures arriving earlier in the year and cold temperatures arriving later.

CONCLUSION

Lotic ecosystems support thousands of species, although their conservation and management are critical, and freshwater biodiversity constitutes a valuable, natural resource. These ecosystems are experiencing declines in biodiversity far greater than those of the most affected terrestrial ecosystems, and the probability to conserve much of the remaining biodiversity in fresh water seems to be very low.

These ecosystems are threatened by many anthropogenic stresses such as overexploitation, water pollution, flow modification, destruction or degradation of habitat, invasion by exotic species, hydropower, and climate change. For instance, cumulative impacts of small dams on biodiversity need to be considered, even when individual installations may have only a small.

Climate change is already affecting the ability of ecosystems to regulate water flows. The regulation of water quality and quantity is a key ecosystem service worldwide. Higher temperatures, changing insolation and cloud cover, and the degradation of ecosystem structure result in the occurrence of more and higher peak-flows on the one hand and, at the same time, impede the ability of ecosystems to regulate water flow. This has major consequences for both ecosystems, with associated species assemblages, and people in the scale of whole catchment areas. In addition to freshwater and wetlands, riverine and alluvial ecosystems, and many forest types are affected by changes in the hydrological regime.

Loss of wetlands due to over-extraction of groundwater, drainage for human uses (reclamation), reduced runoff, and increasing sea level rise, will reduce biodiversity and negatively impact the regulation services of wetlands, such as water purification and flood mitigation (Leadley, 2010).

Biodiversity can play a role in adaptation strategies to both drought and floods through the management of watershed, wetland, forest, and agricultural systems. Maintenance or restoration of forest and wetlands, for example, can reduce run-off in times of floods and also increase water retention during droughts.

Planting trees on slope fields, mini-terracing for soil and moisture conservation, and improved pasture management can also complement actions such as building of small-scale infrastructure in water resources management (CBD, 2009).

The examples provided in this chapter demonstrate also the feasibility of using remote sensing indicators to measure natural variability in wetland extent. Coordinated field campaigns and remote sensing research can yield statistically rigorous relationships between remote sensing indicators and biophysical characteristics of wetland ecosystems. For rivers below the detection limits of remote sensing, watershed modelling within GIS software utilizes remotely sensed land cover information to derive hydrologic and suspended loading information that can be used to infer habitat quality parameters. Larger water bodies may be directly mapped, both in terms of aerial extent and water elevation.

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