Sergi Sabater
Arturo Elosegi (Eds.)

River Conservation

Challenges and Opportunities

Fundación BBVA

River Conservation:
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Edited by

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Fundación BBVA

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Preface

Rivers are an essential part of our landscapes, as silvery lines winding through the land, creating along their path a variety of landscapes that enlighten our spirit. Several facts lie beyond this idealized perception. The first fact is that rivers host a huge variety of organisms that live within, make use of, or transform them, in a complex relationship with their physical template. Another fact is that humans have lived along rivers throughout history, using them for drinking, to harvest fish and other food, for transportation, irrigation or energy production. These two parallel stories, that of functional ecosystems thriving with biodiversity, and that of humans using and modifying these ecosystems, are at the core of this book, since they represent the base of the conflict of river conservation.

River conservation can be defined as a multi-front exercise. One front is raising awareness of the importance of preserving river integrity and biodiversity. The worldwide alteration of river courses through damming, abstraction, and pollution is relatively new for river ecosystems, but also for humans concerned about their preservation. A new paradigm of river conservation is emerging, whose importance is recognized as being similar to biodiversity and to ecosystem functioning. Humans use rivers for multiple purposes, and there are many trade-offs between different uses, which must be taken into account for river management and policy. In addition to these aspects there is also an ethical issue. There are still opportunities to preserve, to use resources rationally, and to protect ecosystems and species; but they require our willingness to do so, and our ability to convey this perspective to the decision makers.

This book is organized around this multi-front perspective. The book never intended to provide an exhaustive account of all different problems affecting rivers, but rather to cover the most important stressors affecting rivers, the ecological responses of rivers, and the potential solutions, from science to policy implementation. The authors of the different chapters are leaders in their respective fields, from hydrology and geomorphology to chemistry, ecology and management. This book represents a distillation of their knowledge, organized with the aim of conserving species, ecosystems, and their functioning. The book is intended for a wide readership, of educated non-specialists, and elaborating on this perspective demanded great efforts from all the contributors. Scholars are very used to writing for peers, but not so used to moving transversally from the original knowledge to a common meeting point, in this case the conservation of rivers. We, as editors of this book, are extremely grateful to this bunch of colleagues who trusted us, and were so patient in following our intuition on how this book should be focused.

The BBVA Foundation has made this book possible, not only by covering all the expenses associated with its production, but specially by trusting our initiative, and by making it possible for all the authors to meet and discuss the book at length. This was made possible by the BBVA Foundation hosting the workshop “River Conservation. Threats, Challenges and Opportunities for a Sustainable Future” at their headquarters in Madrid. During this workshop the embryo of the book was certainly defined. Altogether, the Foundation shows how good sponsorship can be for science, in a country now deeply challenged by the allocation of people and resources. Also, our gratitude goes to the editorial team at Rubes Editorial, for their professional work in the production of the book.

Finally, we extend our gratitude to our families, patient as usual when faced with our crazy ideas, and willing to support the time stolen from our vacations and week-ends. Anyhow, the ultimate goal of this book is to humbly contribute to a better world for our future generations, and this certainly includes our children, and grandchildren!

Sergi Sabater
Arturo Elosegi
April 2013

WHY CONSERVE RIVERS?

Chapter 1

River Conservation: Going against the Flow to Meet Global Challenges

Sergi Sabater, Arturo Elosegi and David Dudgeon

Conservation is a state of health in the land. The land consists of soil, water, plants and animals, but health is more than a sufficiency of these components. It is a state of vigorous self-renewal in each of them and in all collectively. […] Land is an organism and conservation deals with its functional integrity, or health.

Aldo Leopold, 1949

Rivers are among the most diverse and threatened ecosystems on Earth, as they are impacted by increasing human pressures. Because rivers provide essential goods and services, conservation of these ecosystems is a requisite for sustainable development. Therefore, we must seek ways to conserve healthy rivers and to restore degraded ones.

1.1. Global change and implications for freshwater ecosystems

The Earth's human population reached 7 billion people on October 31, 2011 according to the United Nations (2010 Revision of the World Population Prospects), and is projected to rise to 10 billion by 2083. Despite some uncertainties in the precise rate of increase and consequent scenarios of future change, the rapid growth of humans is profoundly altering the Earth system and the biodiversity it supports. The local impacts of anthropogenic activities are not evenly distributed: only 1/8 of all humans live south of the Equator, whereas 50% are concentrated between latitudes 20˚N and 40˚N (Kummu and Varis, 2011) where the landscape and natural habitats have been irreversibly transformed by agriculture and urbanization, and there is intense competition for water resources.

The human footprint upon the planet does not solely depend on the number of people. The per capita use of resources, energy and space have profound impacts on the pressure imposed by a given number of people. For instance, average energy use increased 39% worldwide between 1990 and 2008, resulting in large increases in emissions of CO_2. The largest share of that growth was in the so-called emergent economies (the BRIC countries), where the increase ranged between 70 and 170%. Overall, the European countries, the US, Australia and other large economies are avid consumers of energy, mainly in the form of coal, oil and gas, and hence also make major contributions to greenhouse gas emissions. This extraordinary use of fossil fuel resources is leading to a global transformation of the Earth and its atmosphere and climate that is without precedent.

The impact of humans on the global environment has given rise to the term “Anthropocene”, referring to a new, post-Holocene epoch when planetary changes are driven mainly by human activities. This term was promulgated by the Nobel

Figure 1.1: *A headwater stream in the Spanish Pyrenees. Headwater streams are by far the most abundant type of river in the world and, in total, they harbour a significant proportion of freshwater biodiversity, including many highly-specialised species. Such streams have strong links to the surrounding landscape, with important consequences for habitat conditions and the availability and type of food resources*

Box 1.1

Key concepts in conservation

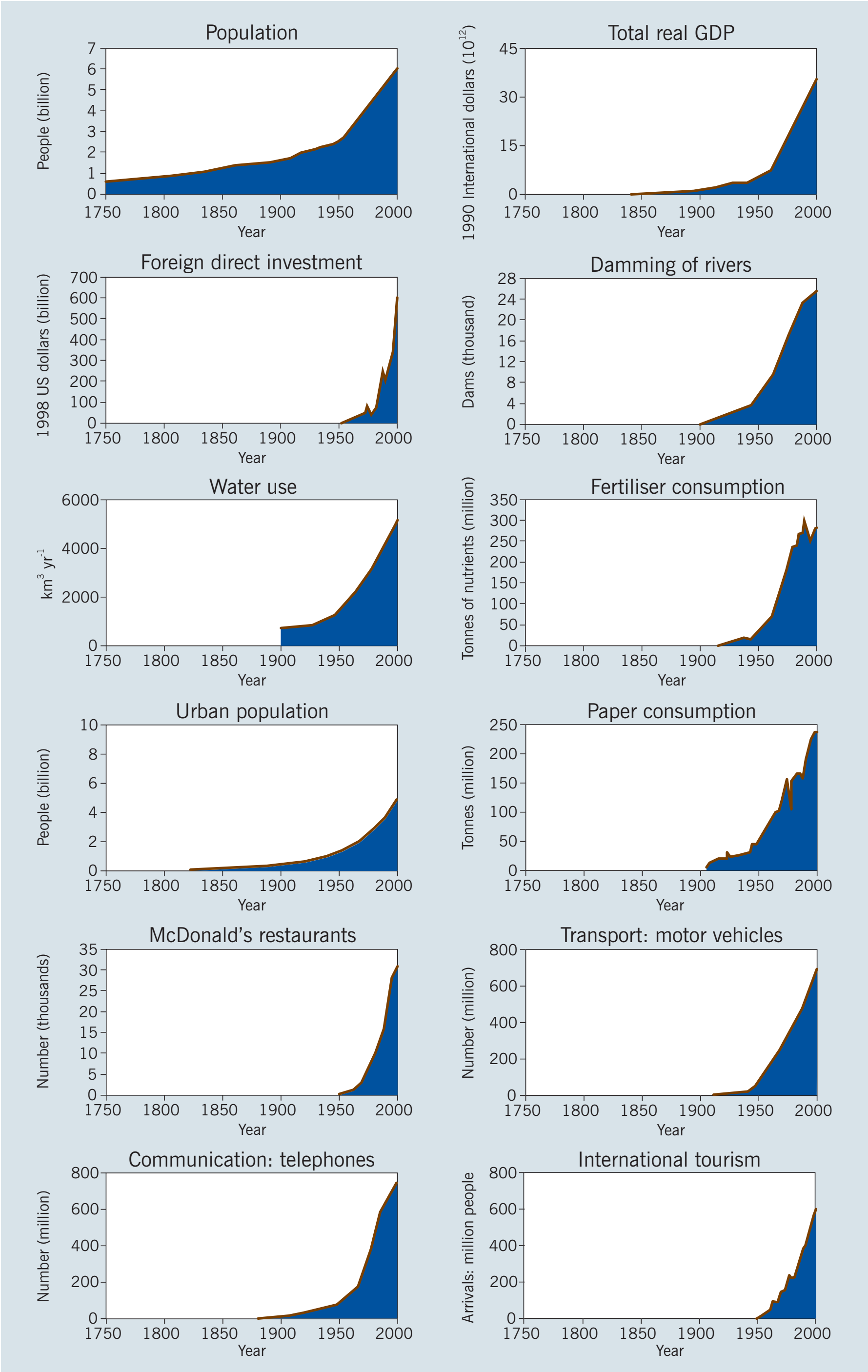
Biodiversity is a contraction of the wording "biological diversity", and thus, refers to the variability among living organisms, including all levels from genes to species and ecosystems. Biodiversity is the result of over 3 billion years of evolution, and ultimately is responsible for many of the characteristics that make this planet habitable, as for instance, the oxygen in the atmosphere, which is produced by plants and other photosynthetic organisms.

Human activities are transforming and degrading natural habitats at an increasing speed, resulting in **biodiversity loss** and often in the destruction of entire ecosystems. Increasing awareness of such problems gave rise to the development of **conservation biology**, or the scientific study of the status of biodiversity with the aim of protecting species, their habitats, and ecosystems from extinction. Conservation biology, thus, is part of a broader movement that aims to conserve nature and the quality of the environment as a way to ensure the well-being of oncoming generations, as well as to protect the intrinsic values of biodiversity.

Nature conservation can operate at different levels. Some actions focus on conserving **populations**, such as preventing overexploitation of a fish species; others focus on the **habitats** where these populations live, for instance, providing gravel beds as spawning areas for fish; still others focus on **ecosystem processes**, such as encouraging the growth of aquatic plants that can enhance water purification.

Prize-winning chemist Paul Crutzen, who argued that the effects of anthropogenic activities on the Earth's atmosphere were sufficiently important to mark the onset of a new geological epoch. While there is some controversy regarding the precise onset of the Anthropocene (some consider it began in the Neolithic period with the invention of agriculture, others associate its origins with the Industrial Revolution), its implications are evident from the ever-growing atmospheric burden of greenhouse gasses (CO_2, N_2O, CH_4…), in the vast amounts of resources appropriated by humans, in the extensive modification of land apportioned to provision of food and other needs, and in changes in the global cycles of water, nutrients and other materials, where the human contributions to the global systems far exceed those attributable to natural processes. As Steffen et al. (2007) have demonstrated, we are in the midst of a phase dubbed "the great acceleration" of population growth and resource use (Figure 1.2); quite evidently, however, this increase cannot proceed indefinitely on a finite planet. The question is not whether it will cease, but when this will happen and what will bring it about. In the immediate future, however, the human ecological footprint and hence the area of land and sea needed to supply the resources we consume and to assimilate our resulting waste, will continue to grow to the detriment of natural habitats and the biodiversity they support.

Figure 1.2: *The dramatically increasing use of resources during the Anthropocene, treated here as marked by the onset of the Industrial Revolution. Global population growth is shown also*



Source: Adapted from Steffen et al. (2007).

The impacts of human activities on the Earth have the potential to be reversible, so long as we do not transgress certain thresholds of sustainability. Unfortunately, some appear to have been crossed already. Of special concern are the effects on biodiversity and on global nitrogen cycling (Rockström et al. 2009). These and other human impacts are not transient and, instead, represent new "baseline settings" for ecosystems and the context in which conservation and management of endangered species must be addressed. Inland waters, and rivers in particular, may be especially vulnerable to Anthropocene impacts because of their strategic position within the global water cycle where they link the atmosphere, soil processes, the biological water in living plants and animals, and the oceans (Meybeck 2003). Maintaining the health of rivers in the Anthropocene world will be challenging: there is hope – but our credit is not unlimited.

1.2. An atlas of global change

While it is indisputable that the global environment is changing, there are large differences between regions regarding the rate, intensity and the nature of change. The tropics, which harbour some of the most diverse and least-known ecosystems in the world, suffer high rates of habitat destruction, often associated with rapid increases in human population. Elsewhere, in parts of North America and Europe, for example, there have even been improvements in the state of the environment, as a result of large investment in environmental policies and implementation of relevant legislation. Note that on a global scale this does not always result in a net gain, since local improvements may be brought about by relocation of highly-impacting activities to countries where environmental standards are less strict.

Regarding rivers, the implementation of the Water Framework Directive in the European Union countries, or the Clean Water Act in the US, represent landmarks in sustainable management, although concerns about water quality remain. Water management is a priority for some developed countries, where human needs for water have been secured in most places. However, water security is often brought about by economic investment in water treatment, rather than by prevention of impacts on freshwater ecosystems, and the result is a gradual depletion of biodiversity (Vörösmarty et al. 2010). In this field the success of legislation, such as the EU Habitat Directive, has been much more limited, because improvements in water quality have not been matched by biodiversity gains.

Elsewhere, however, the effectiveness of technologies is compromised in many countries by lack of enforcement of legislation, widespread corruption, or the tendency to prioritise economic development over environmental protection. Countries with emerging economies often repeat the past errors of states with

Figure 1.3:
A tropical river meanders through the plains of the Amazon basin. Tropical rivers have rich and distinctive biodiversity that, among other things, sustains productive fisheries but, in many places, is currently threatened by ongoing and planned changes to the environment. Lowland rivers are tightly linked to their floodplains, which host rich terrestrial biodiversity

relatively developed economies, with environmental issues ranking low in lists of national priorities.

Because water is a multi-user resource, societal and political interests often become entangled with river management and conservation, as the paradigmatic case of the Iberian Peninsula shows. In particular, parts of Spain and Portugal with a Mediterranean climate tend to experience water scarcity. For instance, in some of the Atlantic catchments water demand is less than 10% of water availability, but the ratio may be as high as 220% in Mediterranean catchments (Sabater 2008). This disparity is maintained by large-scale water transfers between the two regions. Even so, rivers in Mediterranean areas can dry out during extended periods of the year, with dramatic consequences for ecosystems and biodiversity. Thus, the water issue becomes a conflict between human uses and nature conservation, with the usual outcome that consideration about which human uses to satisfy take precedence, whatever the cost for nature. In short, nature does not receive the consideration enjoyed by human stakeholders when it comes to decisions over river water allocation.

The conflict for water for between humans and nature is especially evident in some nations affected by structural deficits. Inadequate management of rivers leads to misery and suffering as a consequence of poor sanitation, water-borne diseases, and flooding. Rural and urban areas often differ greatly in their water availability and safety, which adds to regional inequalities. It will be a significant challenge to meet the legitimate aspirations of growing human populations for a clean, readily-available supply of water, without compromising the water needs of ecosystems and nature.

Irrespective of regional variations, the Earth is changing as a consequence of growing human population and resource consumption. Although there have been a number of advances in nature conservation, many of them are currently threatened by the ongoing global financial crisis, since economic uncertainty tends to push conservation issues down policy agendas. Overall, the annual litany of threatened species added to the IUCN Red List shows the failure of the conservation movement to convince society of the importance of conserving nature, whereas the current economic model drives environmental degradation and loss of natural capital. In the context of a society where consumption is viewed as an essential component of economic "business is usual", nothing is durable and individualism is at stake, preservation of common goods such as biodiversity tend to rank very low in the priorities of most people.

A fundamental change in attitude will be necessary to reverse the Anthropocene trend of environmental degradation and biodiversity loss in inland waters

Figure 1.4:
A highly degraded stream flowing through Fes, Morocco, exemplifies the close relationship between human wellbeing and river health

in general, and rivers in particular. This is what we mean by going against the flow. Thus this book not only concerns facts and figures relating to river ecosystems, it is also a book about human attitudes towards nature, and ultimately a book setting out our responsibility for managing and conserving nature for future generations.

1.3. Why is it important to conserve rivers?

Inland waters are perhaps the most endangered ecosystems on Earth. The decline in freshwater biodiversity is far greater than that in terrestrial or marine ecosystems (Chapter 6), and is attributable to their high species richness in a small area. Dudgeon et al. (2006) report that that 40% of the total fish diversity and one third of global vertebrate diversity (i.e. including amphibians, reptiles and mammals) inhabit freshwater ecosystems, which cover only 0.8% of the Earth surface and represent less than 0.01% of the world's water. Rivers alone constitute an even smaller fraction of this: 0.0002% of all water.

Why is it important to conserve rivers? To the well motivated person, the answer seems obvious: it is in our own best interests, and in the common interest, for us to do so. Nevertheless, despite efforts from the conservation movement, and despite the increasing number of laws and international treaties to conserve nature, the Earth's biodiversity and ecosystems – and rivers in particular – continue to be progressively degraded. While conflicts between humans and nature over water may appear irreconcilable, they are not, simply because healthy rivers are essential to society.

Rivers host a surprisingly large fraction of planetary biodiversity relative to their areal extent and volume, and biodiversity provides the life-support system for humans. Some of the biodiversity values are consumptive, that is, they provide resources that can be exploited: river fisheries, forming the base of many local economies, is an obvious example. River biodiversity is also the source of other cultural and recreational benefits (e.g. tourism) that do not depend on exploitation. More fundamental values of river biodiversity are related to the provision of ecosystem services: water purification, transport and transformation of organic matter and other materials, nutrient cycling, flood control, and others. In any case, the sustainable use of these resources is only possible if we maintain rivers in good health (Box 1.2: see also Chapter 11).

Box 1.2

Key concepts in river science

People tend to think of rivers as one-way pipes that transfer water from land to the sea, or from mountaintops to coastline. This is a gross oversimplification. **Rivers** are **ecosystems** that transport and process water and other materials, dissolved and particulate, organic and inorganic; such materials are derived from river drainage basins or catchments. Rivers are **hierarchically organized**: tributaries merge and create wider, deeper and newer tributaries with higher water volume, and so on. Water travels downstream and therefore effects are also transmitted downstream. This hierarchy also applies to the biological components of the ecosystem. Rivers host diverse **biological communities** distributed within a series of extremely complex **habitats** along the river course and in different parts of its channel. Because of the hierarchical arrangement, biological communities exhibit longitudinal transitions along the river; these are predictable in general terms, but their details depend on specific features of individual rivers. This longitudinal transition of species complement is one reason why rivers sustain so much biodiversity. In turn, this biodiversity constitutes a valuable **resource** for humans, with benefits that include provision of fisheries, shrimps, molluscs and edible plants.

Like all ecosystems, rivers are characterized by their **structure and functioning**. River structure is determined by features such as channel form, current speed, flow volume and water quality, and also by the composition and abundance of the biological communities they harbour (microbes, plants, invertebrates, fishes and so on). Their interactions and combined activities give rise to

Box 1.2 *(cont.)*: Key concepts in river science

river ecosystem functioning, which involves processes such as primary production by plants, transformation of organic materials, nutrient retention, water purification, and secondary production by fishes. In turn, these processes are at the base of **ecosystem services and benefits** of importance for humankind, such as the provision of clean water and flood prevention.

River ecosystems have four important **dimensions** (Ward 1989) that need to be accounted for in management plans:

— **Longitudinal**. Rivers change from source to mouth as they transport materials from the upper to the lower course and throughout the drainage basin. In turn, the physical characteristics of the channel, the range of habitats available and the associated biological communities, change also.
— **Vertical**. Rivers are linked to underground water. An important but often neglected part of the river is the hyporheos, which includes the mass of water circulating among the river sediments, hidden from view, but nevertheless the site of important chemical reactions and biological processes.
— **Lateral**. Rivers are tightly linked to their margins and floodplains, which are integral parts of river ecosystems. When rivers flood large quantities of materials and organisms are exchanged between the aquatic and terrestrial habitats.
— **Temporal**. Rivers are in constant change: in the short term, or on a seasonal basis, periods of rainfall and drought result in dramatic changes in river characteristics, floods disturb biological communities, but at the same time favour the migration of fish and trigger the reproduction of many organisms; in the longer term, river channels migrate laterally or vertically, as the river engineers its floodplain reaches or erodes its valley. Maintenance of this multiscale temporal variability is essential for river health.

Meyer (1997) adds a further relevant dimension: the **social** or **human context**. Rivers are affected directly or indirectly by multiple human activities, and effective river conservation cannot be based solely on the needs of wildlife. Instead, river conservation must take account of the needs and interests of people living along their banks and within the drainage basin.

Most of the early conservation efforts in rivers focused on flagship species of interest for the general public (e.g. salmon), whereas their habitats or ecosystems were neglected. This was problematic, since the decline of a species may be due to slow and or subtle degradation of habitat conditions, and not necessarily a result of over-exploitation or (in the case of migratory fishes such as salmon) dams blocking access to breeding sites. Furthermore, habitat degradation may affect ecosystem functioning, thereby indirectly impacting species of particular interest to humans. Thus, there is an inherent tension in conservation biology: species are the unit for conservation, but the need to provide conditions that allow a species to persist generally requires that conservation efforts focus also on ecosystem protection and/or restoration.

A fundamental question for scientists is how much biodiversity can be lost without seriously compromising natural processes? Although some general principles have emerged (Hooper et al. 2005), there is still much debate on the relationship between biodiversity and ecosystem functioning. The "conventional" view, also called the diversity-stability hypothesis, states that as we lose species, ecosystem function is affected proportionally. A second possibility (the redundancy or rivet hypothesis) is that loss of species has no effect on function until some critical threshold below which ecosystem functioning fails. A third possibility, called the idiosyncratic hypothesis, holds that there are no general rules, that functioning may be unaffected by the loss of certain species, but greatly impacted by the loss of others. According to this hypothesis some species would be more important than others. Amongst these (see Box 1.3) may

Box 1.3

All species are not equal

All species should be viewed as deserving the same level of protection. However, conservation biologists pay special attention to certain species because of their overall significance to the ecosystem as a whole:

Engineer species. Are species that modulate the availability of resources for other species, and so, change the environment creating new habitats. The best known riverine engineer is the beaver, whose dams convert fast-flowing reaches in ponds, flood riparian areas, etc. Other engineers are riparian trees, which create architecturally complex habitats and exert a profound influence on channel form.

Keystone species. Are species that have a disproportionate effect on biological communities, because of their size or activity. Many top predators are keystone species, since their effects on prey cascade down food chains, affecting the entire food web.

Sentinel species. Are species that, because of their sensitivity to changes, can give an early warning of oncoming problems. Thus, scientists can use them like the miners used canaries to detect the existence of toxic gases; typically, they are species with a very low tolerance to pollutants.

Umbrella species. Are species that have a large requirement for space, and therefore protecting them provides protection to many other species. This is the case of the Pacific salmon that helped protect the wood accrual and the river integrity in the Pacific Northwest. Some species may confer this overall protection because they attract public attention **(flagship species)**, as in the case of the platypus in Australia, or dolphins in many tropical rivers.

On the other hand, excessive proliferation of certain exotic **invasive species** is a major threat to biodiversity, as they can displace native species (by completion or predation) thereby tending to homogenize ecosystems throughout the world (Chapter 8).

Figure 1.5: *The Yangtze river dolphin was declared "functionally extinct" in December 2006 after extensive surveys along its home river had failed to yield any sightings. The dolphin has not been reported in the wild since then, and none remain in captivity*

be top predators, which, by feeding disproportionately on certain prey species affect assemblege structure, food webs and ultimately ecosystem functioning, or species that actively modify habitats, like the beaver, whose dams create ponds and reduce the downstream transport of material. A related point is that the magnitude of variability in ecosystem processes increases when species are lost and this tends to reduce the likelihood that multiple ecosystem functions can be sustained (Peter et al. 2011).

One thing is certain: the species cannot be conserved in nature unless we also maintain their habitats and the ecosystems within which they are embedded, plus the linkages between these ecosystems and their surroundings (Chapter 10). This is an especially complex challenge for river networks, which exhibit connectivity in multiple dimensions (lateral, vertical, longitudinal; see Box 1.2) that complicate management and conservation efforts. Therefore, protection of a given species, habitat or river segment cannot be focused on a single location, but needs to include upstream and downstream reaches, the riparian zone or floodplain, and even influences from the entire drainage basin and atmospheric inputs (e.g. nitrogen deposition). A basin-wide perspective for conservation and management is therefore essential (Chapter 12). Further, rivers are dynamic ecosystems, and this often conflicts with management that attempts to conserve rivers as they are now, or as they were in some moment of the past. Since change is an essential characteristic of healthy riverine ecosystems, then conservation and management plans must take account of this changeability and complexity. We know that this is challenging in human-dominated landscapes where the desire for stability or predictability of environmental conditions takes priority over the need to allow rivers to undergo (for example) seasonal flood-pulse cycles (Chapter 2).

1.4. The main threats to river conservation

Rivers offer prime examples of ecosystems threatened by multiple stressors (Figure 1.6), with the interactions between stressors being especially complex because of the hierarchical arrangement and complexity that characterize river systems (see Box 1.2). The effects may be synergistic, and are certain to be further amplified by changes in the global water system driven by climate change, with uncertain – but very likely detrimental – consequences for river ecosystem structure and functioning.

One of the main threats to river ecosystems are alterations of the hydrologic regime (Chapter 2), as they directly affect the availability of water, which is the essential environment for many riverine species and an important force shaping habitats both in the channel and on the floodplain. Rivers, originally characterised by flows that varied with the regional climate, are increasingly affected by either direct impacts such as damming and water abstraction, or indirect ones such as increase of impervious areas in their drainage basin. In most cases the

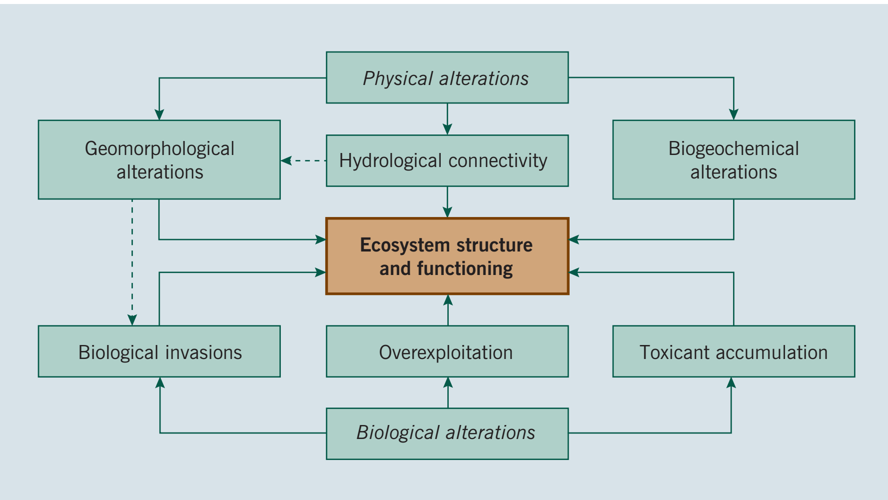


Figure 1.6: *Physical and biological alterations interact through complex feed-backs and affect the biodiversity and functioning of river ecosystems*

frequency, timing and magnitude of floods and droughts are altered, which has detrimental consequences for organisms adapted through natural selection to the natural water regime. In some cases rivers, once wild and noisy, simply cease to flow for long periods, as a result of excessive water drawdown, or are reduced to concrete-lined channels or drains. Such "silenced" rivers are the paradigm of an impaired ecosystem.

Rivers are diverse and dynamic ecosystems but also are among the most endangered habitats in the world. Their degradation reduces biodiversity and ultimately affects human society

Humans have wrought profound changes upon the physical structure of river channels, impairing their life-supporting architecture for riverine communities (Chapter 3). Natural river channels are complex and dynamic, and many species are adapted to and depend upon these characteristics for survival. Even humans have relied on these features, as shown by the example of the ancient Egyptian civilization that depended on the flood cycle of the mighty Nile. Nevertheless, as the technological capacity of humans rose, increasing discomfort with the capricious flooding and wandering of rivers led to huge investments of effort to harness their powers, by damming, building levees, channelizing, or otherwise controlling channel form and mobility. Many rivers are today but a caricature of the complex and dynamic ecosystems they once were, and biodiversity and ecosystem functioning are consequently profoundly compromised.

Pollution is an epidemic and well understood threat to rivers. Sometimes pollution can be caused by natural, even essential substances, when they appear in too high concentrations. This is the case of nutrients, that can be a blessing or a curse, depending on the local conditions (Chapter 4). Nutrients are essential elements for primary producers such as algae and other plants, but when concentrations are too high they result in changes that can lead to fouling of water, lack of oxygen, and declines in biodiversity. Human activities are greatly increasing the amount of nutrients circulating worldwide across biogeochemical cycles, and thus, more and more nutrients tend to reach rivers. This can happen either through diffuse sources, such as atmospheric deposition of nitrogen or nutrients applied as agricultural fertilizers that run-off the land or percolate through the soil into rivers, or through point or end-of-pipe sources such as urban wastewater. Nutrients offer a clear example of the complex relationship between human actions and river health. Some pollutants are more insidious, as they are novel substances, synthesised by humans for various purposes, but which nevertheless end up in rivers. This is the case of pharmaceutical drugs, pesticides, and other substances with potentially-powerful biological effects, which are frequently detected together with their degradation products. These so-called emerging pollutants pose a large challenge to science and environmental management, as very little is known of their action mechanisms in the biota, of their mobility and accumulation in food webs, nor of their interactions in complex mixtures, as can often be found in rivers (Chapter 5). Depending on their chemical characteristics, many pollutants remain within water or are stored in the sediments. Others are more volatile, and thus distributed in the air, can "hop" across watersheds to areas far from where they were originally released. Viewed in this way the entire Earth functions as a huge distiller, with pollutants volatilizing in warm areas and being deposited in cooler localities. The high concentrations of pollutants detected in apparently pristine regions like the Arctic or high-mountain areas show the pervasive effect of human actions, and the need for global responses to the current threats.

Rivers are threatened by changes in the amount and timing of water flow, changes in channel form, excess nutrients and pollution, and spread of invasive species

As a result of these and other drivers including overexploitation, river biodiversity is declining even faster than its terrestrial or marine counterparts (Chapter 6). This places a profound responsibility on the current generation, since societal decisions taken in the next few decades will determine the long-term fate of riverine (and other) biodiversity, and the opportunities (or not) that future human generations will enjoy to appreciate, understand and benefit from that heritage. As species are being lost, serious concerns are raised about the effects of loss of biodiversity on ecosystem functioning – an important and developing field of current scientific research (Chapter 7). Biological invasions are also a cause for serious concern (Box 1.3; see also Chapter 8), as opportunistic exotic species are both promoted by environmental degradation and pose a major threat to native biodiversity. Indeed, the global spread and proliferation of native species may well result in the Anthropocene becoming known also as the Homogocene.

This list of threats to river ecosystems is by no means fully comprehensive. Moreover, global climate change, which may override or magnify, the impacts of some of them, will affect river ecosystems and humans alike. Human adaptation to climatic uncertainty will certainly lead to engineering responses that will further alter river systems globally since, as shown by Vörösmarty et al. (2010), enhanced human water security is generally accompanied by greater pressures upon river ecosystems.

1.5. What needs to be done? Elements for a debate

It is now clear that effective river conservation needs to take a landscape perspective, since rivers depend crucially on processes occurring in their riparian areas (Chapter 9) and on connectivity with the land and across the river network (Chapter 10). This was made clear almost 40 years ago in a seminal paper in which Noel Hynes (1975) stressed the close relationship between the stream and its valley. The relationship between the river and its drainage basin casts the spotlight away from the water alone to conservation of the surrounding land-

Box 1.4

Conservation, an ethical question

One of the current arguments to conserve rivers is that they provide **ecosystem services**: i.e. goods and other benefits that are important to society. These benefits decrease or can even disappear when river ecosystems are degraded. Assuming this is correct, it can be a powerful argument for conservation, especially if we are able to value these services in economic terms. Evidence that services produce economic benefits would provide a strong incentive for decision-makers and managers to implement effective conservation measures for river ecosystems. The relationship between nature conservation and ecosystem services is not straightforward, and matters such as inter-generational value (or inheritance value) of biodiversity may not be easily monetarised.

The question remains of whether we should value species or ecosystems only inasmuch as they benefit us humans directly. This is not a scientific question, but an ethical one, and scientists are no better equipped than anyone else to provide the right answer. Nonetheless, ethical questions are an important part of the human dimension of river conservation. Humans, by their very existence, modify the environment, and thus, a large human population will always have an impact on nature. We cannot avoid exploiting natural resources and transforming the landscape to some degree. The value of nature is not determined solely by the satisfaction it provides for human needs, since species and ecosystems that evolved through millennia cannot be defined purely in terms of their utility for a relatively recently-evolved sentient ape. Instead, these products of evolution have their own intrinsic value – in and of themselves.

This ethical issue can be put as follows: species have their own right to exist and, if that is the case (or even if it is not) humans have the obligation to pass on to our descendants an intact version of the diverse biosphere that we ourselves have been fortunate enough to enjoy. *Homo sapiens* is the only species on our planet capable of pondering and acting upon ethical concerns about the species with which we share both a planet and an evolutionary history, and it behoves us to act in a way that avoids driving them to extinction.

scape, this is by no means a simple matter. River ecosystems face many threats, as do many humans dependent on scarce, unpredictable and/or contaminated water sources. Pressures on rivers from burgeoning human populations, their need for drinking water, for water for agriculture and industry, are all growing, and the looming spectre of anthropogenic climate change adds uncertainty as to how pressure on water resources may change in the medium and longer term. We have already lost some significant components of freshwater biodiversity, but far more are in decline. Nonetheless, appropriate action now has the potential to make a large difference for future generations. This is a time for action: the present generation is likely the last with a chance to preserve a significant fraction of the biodiversity inherited from our ancestors and thereby bequeath it to our descendants.

Although species that have become extinct are gone forever, part of the damage done to freshwater ecosystems is potentially reversible (Chapter 11). Visible signs of this include the dramatic improvement of water quality in many European rivers (Tockner et al. 2009), and the growth of citizen stewardship related to river conservation in several countries (Chapter 13). Rivers are especially resilient ecosystems: pollutants tend to be diluted and washed downstream much faster than they disappear from soils, and river biota, shaped by natural selection in a highly variable environment, shows remarkable resilience and, so long as connectivity within rivers is maintained, rapid re-colonisation ability. Recovery of degraded rivers also requires alleviation of human stressors or pressures upon them, which demands both scientific understanding and, perhaps more importantly, political will.

One possible avenue for effective conservation is to increase efforts in river restoration. Ecosystem restoration is an activity devoted to recovering lost structure and functions, and thus, should not be confounded with “gardening” river margins in urban areas, as it often is. Instead, river restoration should focus on recovering the dynamic characteristics of rivers, characteristics that include the channel lateral mobility and the capacity to flood the floodplains, upon which many important ecological features and ecosystem functions depend (Chapter 11). But to what state should a river be restored? In some parts of the world the answer can be straightforward, but in regions such as Europe with an environment substantially modified by humans for centuries, the question is not trivial. Should we restore a river to its state 100 years ago? To the pre-industrial state? To its state in the Middle Ages? As we go further back in time, we have less precise information on the state of the river, and must face the possibility that land use transformation, the establishment of non-native or invasive species, and climate shifts may make it impossible to restore the river to its original state. In such circumstances, it may be more appropriate to aim at river rehabilitation: i.e. changing the condition of the river to an extent that some ecological functionality can be maintained and some enhancement of biodiversity brought about.

Irrespective of whether the goal is river restoration or rehabilitation, conservation and management practices must be integrated within a landscape framework. As mentioned above, river conditions are the product of activities within their drainage basins, and the solution to problems at one locality within the river network often lies some distance away within the catchment or upstream.

The landscape framework is necessary not only when addressing pollutants originating in the drainage basin. Rivers are non-linear ecosystems that form branched networks through which water and material are transported longitudinally. This architecture imposes special constraints upon the movement of aquatic animals, which has important implications for the long-term viability of populations. For instance, when any factor causes the loss of a given species in a particular reach, the population of this species can sometimes recover if there are upstream sources of colonists. Populations inhabiting other streams in the same network can also be a source of colonists, but only if there are no barriers (such as dams or waterfalls) to dispersal from the source to the receiving reach. Therefore, this type of barriers can produce far-reaching impacts by blocking animal migration. A broader perspective is needed when considering the loss of a species from an entire drainage since, in this case, recolonization will only occur by way of dispersal across a terrestrial landscape (possible for dragonflies, not so for fish) or by fully aquatic animals travelling along the coast between river mouths. This can only occur if the species concerned has tolerance for saline water. For species that can neither travel across land nor tolerate salt, there may be a case for human intervention to reintroduce native species to degraded rivers so as to facilitate restoration or rehabilitation efforts.

The present environmental crisis cannot be attributed to a lack of knowledge. Indeed, it could be argued that it is rather a product of a failure to apply the knowledge that already exists. Nevertheless, many scientific questions remain to be addressed. There is an urgent need to gain knowledge of the effects and fate of the many toxic compounds (pharmaceuticals, cosmetics, and a host of pesticides and other chemicals) that are being added to inland waters globally, so as to better understand their effects on humans and ecosystems (Chapter 5). We need better knowledge of the transport and fate of pollutants in the biosphere, of their bioaccumulation and of their interactions. We also need deeper understanding of biodiversity and its relationship to ecosystem functioning and the benefits enjoyed by humans since, at present, the function and ecological role of the vast majority of species remains unknown or uncertain (Chapter 6). This is particularly so for microorganisms. An entirely different challenge is posed by the need to enhance river restoration and rehabilitation efforts: we must define ways to restore the dynamism of fluvial channels, so as to provide the appropriate conditions to maintain biodiversity and ecosystem functioning, and to do so in a highly modified human-dominated landscape. Maintaining the spatiotemporal variability of river ecosystems (e.g. the annual flood cycle), to which the native biota are adapted, provides one of the best defences against the invasion by exotic or non-native species. It is also necessary to find ways of connecting populations of animals that have been fragmented by dams or by highly-altered river reaches. Environmental water

Box 1.5

Sustainable development; what is it?

Sustainable development is a widely-used term, but one that is often applied in very loose, even contrasting ways. **Sustainability** implies use of resources in a manner that does not restrict the opportunities of future generations to use them.

Sustainability can be seen as a three-legged stool with **social**, **economical** and **ecological** legs. The social leg is linked to the goals we desire as a society; the economic leg refers to the ways we devise to reach these goals; and the ecological leg concerns the limits imposed by nature. Each leg is essential to the stability of the stool; in particular, the stool will collapse if the ecological limits of the system are exceeded and that leg is no longer able to support the whole.

Management of fisheries offers a good example. Societies must decide the needs to be met when exploiting a fishery; this might include goals such as deriving the highest possible revenue in the long term, or combining fish production with conservation of leisure opportunities. To achieve this, resources must be used in an economically sound way, by investing money in renewing the fishing boats or establishing size limits. Whatever decisions are made, they must take account of the ecological context, the biological capacity of the fish population to be harvested. Many fisheries have been doomed as a result of political decisions (on economic or societal grounds) to exploit more than the sustainable limits proposed by scientists, or simple inaction and failure to introduce legislation to control overfishing. In the long term, of course, decisions that fly in the face of ecological reality are hardly likely to benefit societies or the fishery stocks they (over)exploit.

allocations (e-flows) for rivers must also be defined so as to meet the needs of intact ecosystems while, at the same time, satisfying human needs for water. This will be a major challenge in more arid regions (for example) where water that is allowed to flow to the sea may be regarded by human users as “wasted”. Again, the need to achieve this balance highlights the necessity of gaining a better understanding of the relationship between biodiversity and ecosystem functioning.

Above all, it is essential to educate citizens and to demonstrate and convey the importance of freshwater biodiversity to ecosystem functioning as well as the need to protect both that diversity and the benefits accruing to humans. Arguably, awareness is the key thing for society; once we become aware, then we will force governments to proceed with conservation action and embed such action in appropriate policies. To date, however, it seems that public perception of the importance of freshwater biodiversity, and the need to protect it, falls far short of what conservation biologists wish for, and fuller engagement of the scientific community will be needed to translate our good intentions into the necessary action by public stakeholder groups. In the future, our generation will be judged, not by how we tackled the financial crisis, nor by the beauty of our architecture, but by the biological heritage we leave, and especially by the state of the environment. Success in that regard will require an innovative combination of scientific knowledge, political conscience and economic perspicacity with societal willingness. We must hope it is not too late to bring this about.

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STRESSORS AFFECTING RIVERS

Chapter 2

The Silent River: The Hydrological Basis for River Conservation

Tim Burt

We may conclude that in every respect the valley rules the stream.

Hynes, 1975

From headwaters to mouth, the physical variables within a river system present a continuous gradient of physical conditions. This gradient should elicit a series of responses within the constituent populations resulting in a continuum of biotic adjustments and consistent patterns of loading, transport, utilization, and storage of organic matter along the length of a river.

Vannote et al., 1980

Rivers, as hydrological systems, have a highly integrated nature: changes anywhere in the catchment can have significant effect further downstream. Rivers differ in their hydrological regimes, following general regional patterns. These regimes are important as they provide a context within which human influence on the river system can be defined. The pathways followed by the water to reach the channel determine its sediment and solute load. Therefore, climate, land use change and channel engineering can impact river hydrology along with the prospects for delivering sustainable water supply.

2.1. The river continuum

People are constantly changing entire landscapes, deeply affecting river ecosystems. Many of the key impacts are linked to water. Some of these changes are deliberate: building dams, diverting water for irrigation, taking water for domestic and industrial use. But some impacts are indirect, and yet equally influential. These include major changes in land use like urbanisation or deforestation, which can change the whole character of the river as well as the surrounding land. Many of these impacts result in reduced flow in the river: there may be fewer flood peaks but also lower flows as well. Water withdrawal results in domesticated, silent rivers where most of the natural functions have disappeared.

The river basin is an open system with outputs from upstream areas (land surface and channels) providing inputs to downstream sections

Today, there is an intuitive assumption that the condition of the river channel and its drainage basin are intimately linked, but this was not always so. A series of research papers in the 1970s encouraged a new approach to aquatic ecology, borrowing concepts from fluvial geomorphology that stressed the drainage basin as the fundamental unit of analysis. This concept is known as the *river continuum* concept (Vannote et al. 1980; see review in Burt et al. 2010). This takes the view that, at any point on the river network, there is a balance between the water moving through the channel and the resulting habitat and species mix. The river is regarded as an open system, with the output from one section providing the input to the next. Hence, the river network can be seen as an integrated system, with a clear connection between upstream and downstream. At any point along the channel, the amount of water flowing in the river, its chemistry and sediment load, all reflect processes operating within the *entire* river basin upstream of that point (i.e. not just in-channel conditions). Moving from source to mouth, the in-stream biological community is constantly adjusting in response to progressive downstream changes in discharge, energy inputs and nutrient availability.

At the same time as the river continuum concept was proposed, another group of researchers developed the *nutrient spiralling* concept (Webster and Patten 1979) which describes how, as organic matter and nutrients flow downstream, they are taken up by plants, and then perhaps eaten by animals. Later, the plants and animals die off and matter is released back into the river water. The nutrients seem to "spiral" along the river, from the water to the biota and *vice versa*, constantly being taken up and released. Together, these two concepts underpin our current understanding of river ecology, emphasising production, cycling and transfer of energy and organic matter along the stream network. We can easily forget just how vital water is to river ecology; we must remember always that any changes to the quantity or quality of river flow are bound to have major consequences.

Another key idea is the importance of temporal variability in discharge for river ecology. Each type of river has its own natural flow regime, to which the local fauna tends to be adapted, and thus, any change of the flow regime is likely to result in changes in the biological communities (Poff et al. 1997). Important points in the natural flow regime are the number of flood events or spates per year, the time the river requires to return to base flow after the spate, and the predictability of spates, this is, the extent to which spates occur at the same time every year. Small streams tend to be very flashy, i.e. discharge increases swiftly following rainfall, and recedes again rapidly to base flow. Large rivers, on the other hand, because they receive the flow from vast drainage basins, tend to have more slowly rising and receding flows, and also more repeatable hydrographs from year to year. In the largest rivers, floods can last for months and flood vast areas of the floodplain; this flood-pulse is a key driver of the ecology of large rivers (Junk et al. 1989). For instance, many fish species breed in flooded terrestrial habitats which get nutrients from the floodwater, nutrients that can later return to the river channel in the form of fruits, leaves, and other organic materials. Additionally, floods are important for many migratory fishes, as features such as small chutes, debris dams or shallow areas, that are impassable barriers during low flows, can be easily traversed during floods. Therefore, the flow regime, the natural alternation of floods and periods of low discharge, is an essential element of river health.

Traditional river engineering provided solutions at a particular site, often with little or no regard being paid to upstream conditions. Today, the drainage basin is viewed as a single, connected system. This requires an integrated approach in which upstream conditions, both in the channel *and* in the catchment area surrounding the channel network, are fully taken into account. This spatially "distributed" approach focuses our attention on the sources of water, sediment and solutes being transported within the river channel. Moreover, the pathways by which water and any material being carried in the water ("load" – a mixture of solid and dissolved material) must also be understood, so that connectivity between "source" and "target" can be fully appreciated.

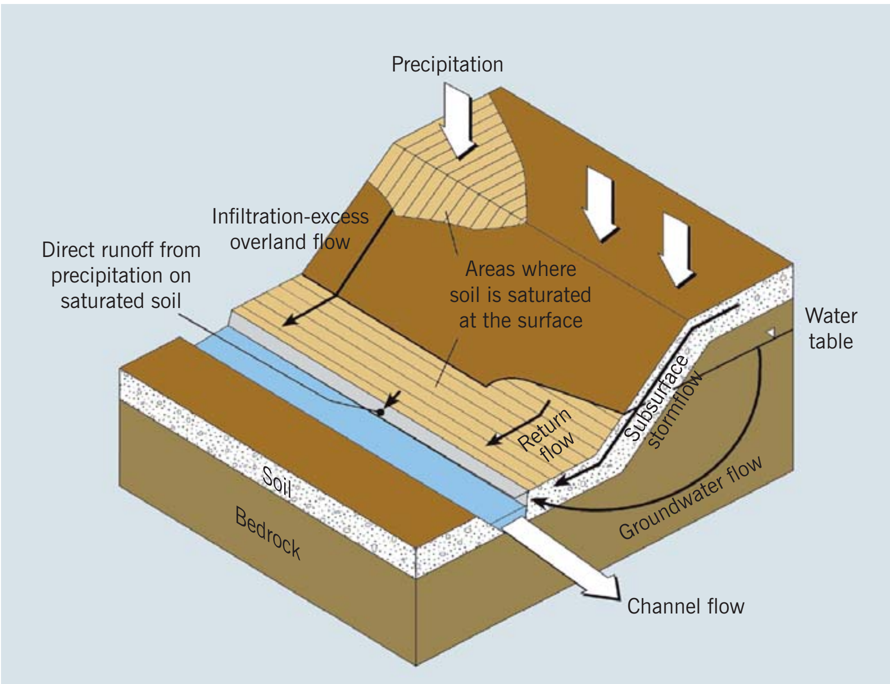
Any changes to the quantity or quality of river flow are bound to have consequences for river ecology

2.2. Where does the water go when it rains?

The nature of the soil and bedrock determine the pathways by which runoff from the catchment area will reach a stream channel. These flow paths determine the speed and volume of water travelling to the nearest channel and the load of sediments and dissolved substances acquired by the moving water. Runoff pathways are studied using a hydrograph, a plot of discharge (with units of volume per unit time) against time. For very small basins, it is necessary to plot instantaneous discharge values whereas for larger basins, mean hourly or mean daily discharge values will suffice.

The water falling during a storm can follow different pathways, depending on the local conditions (Figure 2.1). Part of the water is intercepted by the vegetation, part is evaporated again directly to the atmosphere, and part of the water reaches the soil surface, where it tends to infiltrate at a rate that depends on the slope, porosity, and moisture content of the soil. When the rainfall intensity exceeds the infiltration capacity of the soil, water flows over the land surface, where it can cause erosion, especially if the soil is bare (Figure 2.2). When it rains on a ground that is already saturated, all the water must flow overland. Often the water infiltrates the soil, but reaches an area that is already saturated with water or which is less permeable than the topsoil, and thus, water emerges from the ground and flows across the surface. Water flowing through the soil (sometimes called *throughflow)* moves much more slowly than overland flow, but the response can still be quite rapid when newly infiltrated water shunts water that was already in the soil out of it and into the stream, or when the water moves through large cracks in the soil rather than through the soil matrix. Water that percolates into the bedrock moves at much lower velocities by longer flow paths and takes much longer to reach the

Figure 2.1:
Block diagram of hillslope runoff processes showing the main pathways followed by rainfall



Source: Burt (1992).

Figure 2.2: *Soil erosion on an agricultural field in southern England caused by infiltration-excess overland flow*

stream channel. The long residence time of groundwater usually means that it has a much higher solute concentration than overland flow, simply because this gives longer for rock material to dissolve.

Figure 2.3 shows a hydrograph for a small headwater basin in south-west England; also shown is the solute concentration of the stream water. This flood hydrograph was generated by an intense storm of 25 mm rain in just 15 minutes. Note how the stream water is rapidly diluted by input of overland flow. The storm hydrograph has been divided into "new" and "old" water. The new water – overland flow – reaches the channel quicker than the old water which has to move through the soil. In this case, the flood is quickly over and the stream returns to baseflow conditions – low flow with a higher solute concentration than the storm runoff.

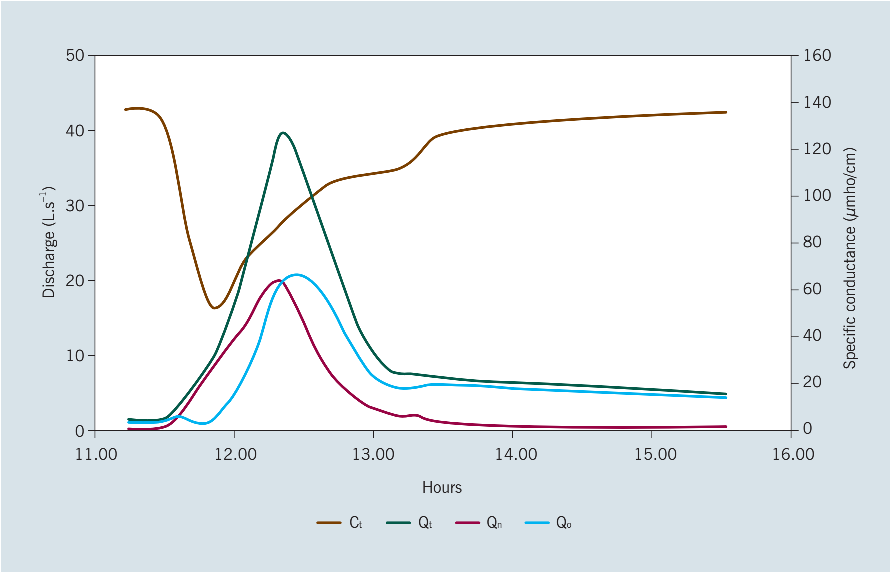
2.3. Water balance

The water balance for a river basin over a selected time period (usually annual) can be evaluated as follows:

$$P - Q - E - C - \Delta S = 0$$

where P is precipitation, Q is river discharge, E is evaporation, C is water consumption (that is, the amount abstracted by the human population and not

Figure 2.3: *Storm discharge (Q_t) and stream solute concentration (C_t) for the Bicknoller Combe stream. Old (Q_o) and new (Q_n) water contributions estimated using a chemical mixing model (adapted from Burt, 1979). Specific conductance is a proxy for total dissolved solids concentration*



returned to the river; Milliman and Farnsworth 2011), and ΔS is change in storage. In some river basins there may be leakage of groundwater into or out of the basin (as defined using surface topography) but in most cases this can be ignored.

By way of example, the mean water budget for a small catchment in south-west England (Slapton Wood) over a 37-year period was as follows:

$$P = 1066 \text{ mm}$$
$$Q = 540 \text{ mm}$$
$$E = 524 \text{ mm}$$

Note that 48% of rainfall was converted into river flow and 52% was lost through evaporation, a very typical result for a lowland basin in a warm temperate climate.

2.4. Global hydrology: Climate and river regimes

The regime of a river may be defined as the seasonal variation in its flow and is usually portrayed by a curve based on monthly mean flow (Burt 1992). Seasonal variations in the natural runoff regime of a drainage basin depend primarily on climate. Of course, vegetation cover plays an important role too, but it is also controlled by climate and so not a truly independent driver. As noted above, soil and bedrock control the rapidity of runoff: impermeable soils encourage rapid storm runoff response; basins with permeable soils and deep aquifers will greatly attenuate the link between rainfall and runoff.

Beckinsale (1969) noted that river regimes in most parts of the world reflect the regional climatic rhythm. Therefore, he modified the classical classification of world climates made by the German climatologist, Köppen, and so was able to produce a generalized delineation of hydrological regions (Figure 2.4). These are the main types of hydrological climate according to Beckinsale:

A = tropical rainy climates
B = dry climates with an excess of potential evaporation over precipitation
C = warm, temperate rainy climates
D = seasonally cold, snowy climates

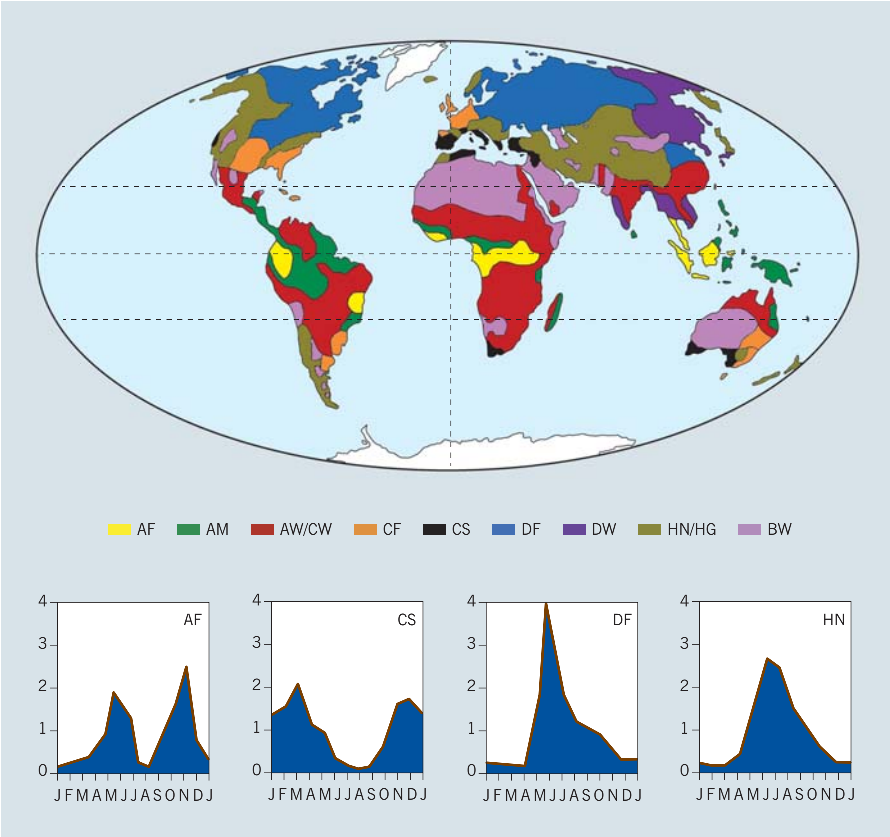
Beckinsale applied the rainfall symbols of Köppen to provide the second capital letter in the code:

F = appreciable runoff all year
W = marked winter low flow
S = marked summer low flow
M = moderate low-water season

Beckinsale added a further class to take into account regimes that occur in the snow and ice environments of high mountains (H) outside the polar ice caps: HN and HG, denoting nival (N, dominated by spring snowmelt) and glacial (G, dominated by summer glacier melt) regimes respectively.

Generalized classifications like that of Beckinsale (1969) remain useful as a context for more local analysis. Recently, Milliman and Farnsworth (2011) have used temperature, total annual runoff and season of *maximum* runoff as the basis for their classification of global river regimes. Maximum runoff is more meaningful for considerations of stream sediment transport to the oceans, as most sediments are transported during floods (Chapter 3). On the other hand, minimum flow is relevant to river ecology because in-stream biota are stressed by low flows, which is why the Beckinsale scheme is retained here. Not surprisingly, the data base of river discharge now available is very much more extensive than it was in the 1960s. On the other hand, many more rivers are affected by abstractions so that far fewer regimes remain "natural" and river biota must adapt accordingly to the modified flow regime.

Figure 2.4: *The world distribution of characteristic river regimes with type examples from four regions. Letters are specified in the text. The four rivers shown are: Lobaye, Congo (AF); Volga (DF); Arno (CS), Rheus, Switzerland (HN). Data plotted for the four rivers show the ratio of mean monthly flow to mean annual flow*



Source: Adapted from Beckinsale (1969).

2.5. Flow regimes at the regional scale

Once we begin to examine river regimes at the regional scale, the relative importance of climatic variation diminishes somewhat and other drivers increase in significance. Figure 2.5 shows regimes for two of the largest UK rivers: the Tay and the Thames. The headwaters of the Tay are in the rainy Scottish mountains so the Tay has much higher absolute flows in every month of the year, compared to the Thames. The regimes shown confirm Beckinsale's classification (CS) for UK rivers: a warm, temperate rainy climate with a marked summer low flow.

Drainage basins dominated by surface runoff respond rapidly to precipitation or snowmelt, whereas groundwater-dominated basins have a dampened response, with a greater delay between input and output. This tends to result

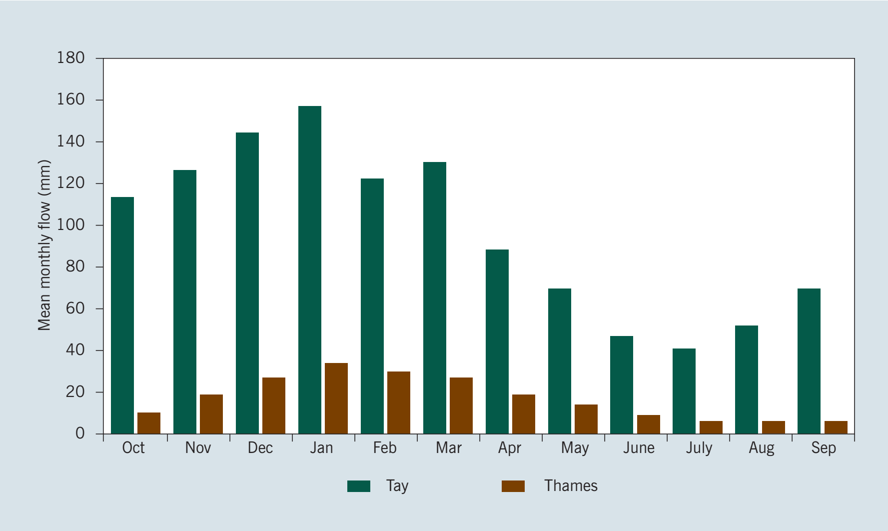


Figure 2.5: *Mean monthly runoff (mm) for the rivers Tay and Thames, UK*

Note: Data from UK National River Flow Archive (station numbers 15006 and 39001).

in a more uniform runoff response compared to the more "flashy" response where surface runoff dominates. The influence of geology on the different water regimes can be exemplified by the Ock and Dun rivers, two tributaries of the River Thames in southern England that, despite having almost identical annual runoff (Ock 211 mm; Dun 222 mm), have very different hydrographs (Figure 2.6). In the Ock, a basin with impermeable clay soils and a dense network of ditches and drains, there is a rapid response of the river to rainfall, with intense, short-lived flood peaks, but very low baseflow in rain-free periods. In the Dun basin, the substrate is permeable limestone, and thus, little storm runoff is produced despite the steeper terrain, and groundwater provides almost all the streamflow. Note that these hydrographs include the severe drought of 1975-76; the complete lack of storm runoff during the very dry winter of 1975-76 is starkly evident.

2.6. Climate change and long-term change in river flow response

The natural flow regime is by no means constant from year to year. We expect catchment hydrology to vary as a direct result of climatic variability (e.g. Figure 2.5). But what if there is long-term climate change? This must gradually affect the response of the river basin. In terms of temperature change, if rain tends to fall instead of snow, this might alter the timing of runoff, with more winter floods and lower runoff from snowmelt in the early spring.

Figure 2.6: *Hydrograph for two contrasting English rivers: River Ock and River Dun. The rain-fed Ock is flashier, has more peak flows and lower base flow than the groundwater-fed Dun*

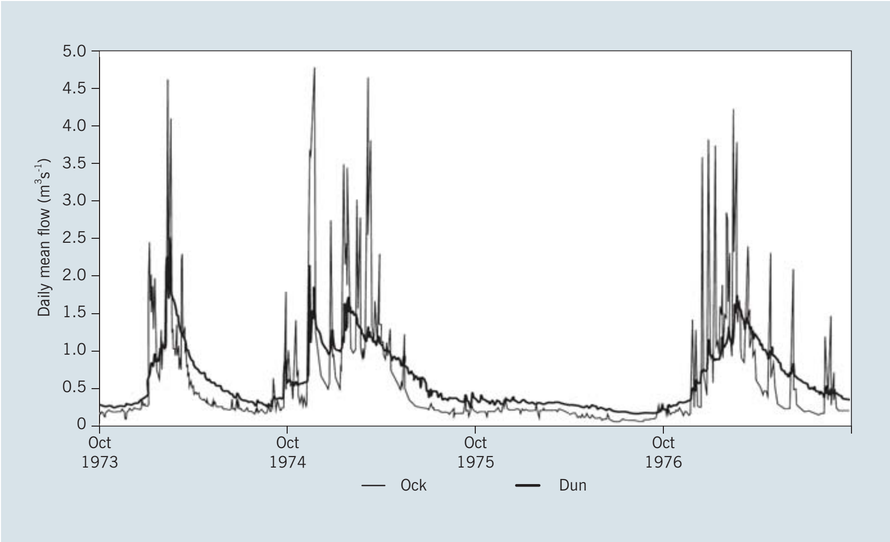


Figure 2.7 shows the relationship between the North Atlantic Oscillation Index (NAOI) and seasonal river flow totals for two UK rivers. The NAOI is a measure of the strength of the North Atlantic circulation. A highly positive NAOI means strong winds and many rain-bearing weather systems crossing north-west Europe. In the River Tay there is a strong positive correlation with the wintertime value of NAOI and it is clear that when atmospheric circulation in the North Atlantic is at its strongest (highest values of NAOI), the winter is likely to be very wet, with implications for flooding. On the other hand, when the NAOI is highly negative, Scotland is likely to be dominated by "blocking" high pressure systems over Scandinavia and winter rainfall will be very low. On the other hand, in the River Derwent, in north-eastern England, there is a negative correlation with the summertime value of NAOI. In this case, when NAOI is highly positive, river flows will be very low, with drought threatening in-stream biota, especially in rivers where there are large water abstractions for domestic supply or irrigation. If the climate were to change so that, for example, the NAO becomes more positive, this would have long-term implications for river ecology: increased winter flooding could destabilize channel systems but low flows would become more common in summer, threatening the viability of some species and ecosystems. Given the tendency of the NAO to fluctuate considerably in just a few months, the most worrying sequence as far as low flows are concerned are two dry summers separated by a dry winter. This happened across England and Wales in 1975-76, producing extreme drought conditions (see Figure 2.5). Box 2.1 provides another example of changes in water resources related to inter-annual climatic variation.

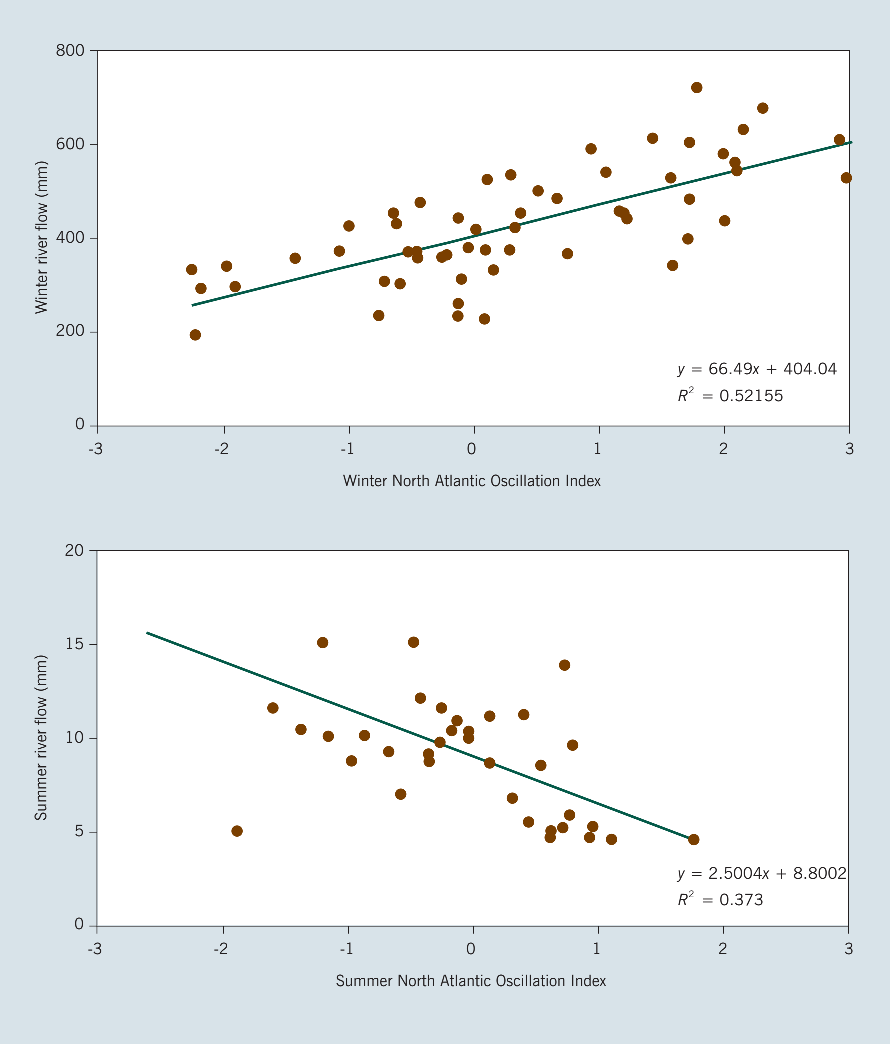


Figure 2.7: *Relationship between wintertime NAOI and flow in UK rivers.* (Up) *River Tay, Scotland.* (Down) *River Derwent, England*

Note: Data from UK National River Flow Archive (station numbers 15006 and 27041).

2.7. Impacts of human actions on river flows

In addition to those caused by climatic variation, human impacts also have long-term effects in river flows. As mentioned above, human impacts may be either *direct*, such as those derived from reservoir construction or water abstraction, and *indirect*, produced by changes in flow pathways across the drainage basin, for instance, because of impacts related to the condition of the land surface are probably more important: these include agricultural practices, deforestation or urbanisation.

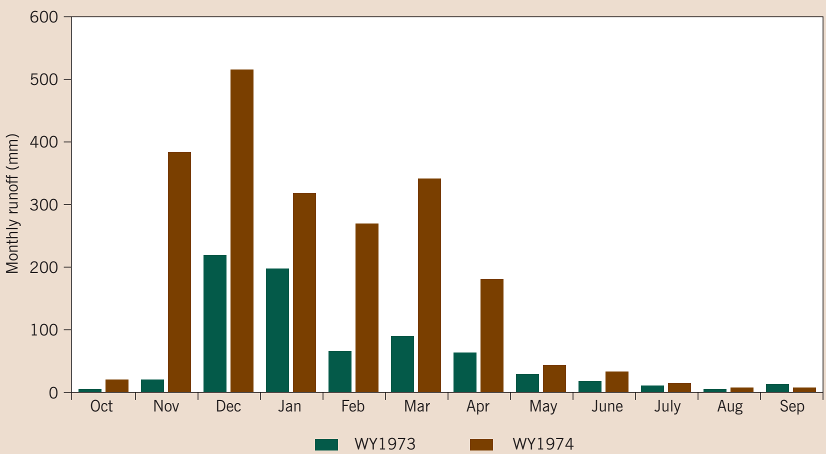
Box 2.1

River regimes and climatic variability

The river regimes presented in sections 2 and 3 are calculated for mean conditions; thus, climatic variability is disregarded. In most places, this simply means ignoring modest differences between wet and dry years, but in some places, variations in ocean-atmosphere coupling can result in very large changes in the pattern of river flow. The figure 2.8 shows the runoff regime at Watershed 2 at the H J Andrews Experimental Forest, Oregon, USA. The water year (WY) runs from 1st October the previous calendar year to 30th September. This basin has a highly seasonal climate with winter snow and a marked summer minimum: Beckinsale maps this region as HN.

First, a brief explanation of the El Niño Southern Oscillation (ENSO) is needed. El Niño (EN) conditions in the Pacific Ocean are characterized by a large-scale weakening of the trade winds and warming of the surface layers in the eastern and central equatorial Pacific Ocean. El Niño events are accompanied by swings in the Southern Oscillation (SO): the pressure gradient along the Equator reverses which in turn reverses wind direction. Instead of cold, deep water upwelling off the coast of Ecuador and Northern Peru, warmer surface water is blown from the west, causing an increase in rainfall in the otherwise arid eastern Pacific. Meanwhile, there is drought in the western Pacific over Indonesia. La Niña, the reverse phenomenon, is associated with a larger than normal pressure difference between the western and eastern Pacific, resulting in stronger than normal trade winds so that upwelling is significantly enhanced off the coast of South America. It is very wet in the western Pacific. These shifts in the condition of the Pacific Ocean, which happen typically every 2-7 years, exert strong control on the climate of the continents surrounding the Pacific Ocean.

Figure 2.8: *Monthly flows (mm) for WS2 at the H J Andrews Experimental Forest, Oregon*



Source: CLIMDB/HYDROBD Database.

At H J Andrews, the very wet year (WY 1974) was affected by La Niña conditions whereas the previous year (WY 1973) experienced a drought as a result of an El Niño event. Whilst the overall pattern of runoff remains the same, the largest differences are seen at the beginning and end of the wet season, November in particular. Although the increase in low flows is relatively small (+31% in July, +68% in August), these have considerable implications for river ecology since headwater tributaries may dry up in El Niño events.

The first recorded dam was constructed in Egypt some 5,000 years ago. Since then, they have been built everywhere except Antarctica as people seek to improve crop yields, prevent floods, generate power and provide a reliable source of water. There are, for example, some 750,000 dams in the USA alone; paradoxically, most are small, but just 3% of the total account for 63% of the total storage volume (Goudie 2006). The main period of construction was during the second half of the twentieth century, and whilst there are still plans to build more large dams, for example along the Mekong River in China, in some places dams are being removed, in recognition of their adverse impact on river ecology (see, for example, http://or.water.usgs.gov/projs_dir/marmot/index.html). Whilst the river continuum continuity is a valid paradigm for river systems (see section 2.1), few rivers in the world are completely unaffected by the presence of dams. Dams completely truncate the channel network, and introduce the so-called *serial discontinuity* (Ward and Stanford 1979), that is the disruption of the channel continuum in hydrological, geomorphological, biological and biogeochemical terms. The disruption relates both to the interruption of the movement of water and the associated load, as well as to the replacement of shallow, flowing water *(lotic)* by relatively still, deep water *(lentic)*. The impact of the dam on the river downstream of the impoundment depends on the size of the dam and the extent to which unaffected tributaries join the affected river. Sediment retention behind a dam is a particular problem, reducing the amount of flood-deposited nutrients on floodplains and causing clear-water erosion of the channel downstream of the dam (Chapter 3).

The impact of reservoirs on the water balance includes abstraction of water for domestic and industrial purposes; some of this water may be returned to the river, but large quantities may be piped out of the basin altogether. Mean annual water balance calculations assume that there is no long-term change in the average amount of water stored in the soil and bedrock, additions in wet years being balanced by losses in dry years. However, this may not always be the case. In many countries, groundwater is being "mined" in unsustainable quantities, resulting in

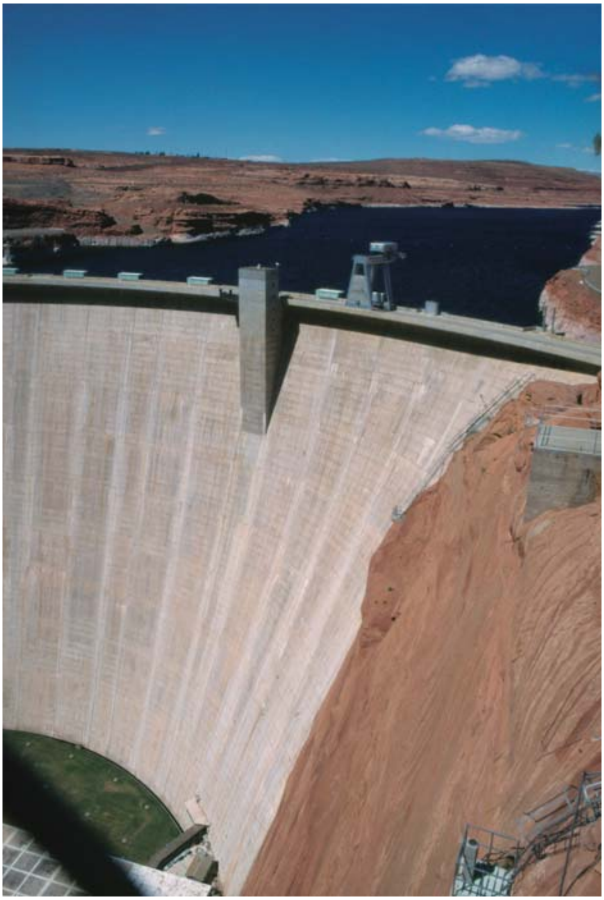


Figure 2.9:
The Glen Canyon Dam on the Colorado River provides a good example of problems that arise when a river is completely blocked, including alterations to the flow and thermal regime of the river downstream of the dam, threats to endemic fish species, and drowning of scenic landscape. Benefits were overvalued and costs underestimated. It is ironic that the Lake Powell is named after the man who first drew up recommendations for the sustainable use of water resources in the arid southwest USA

falling water tables and headwater tributaries drying up (Pearce 2007). For example, groundwater tables have fallen by up to 100 m in the Las Vegas area since the early twentieth century, in response to increasing demand. Note that water abstracted for irrigation is mainly lost through evaporation and does not return to the river therefore. The most devastating impact of irrigation has been the demise of the Aral Sea, which covered over 68,000 km^2 in 1960, and was reduced to a mere 10% of its original size by 2007. The demise of the mighty Aral resulted in huge impacts both on biodiversity and on human populations. The case of Moynaq in Uzbekistan, formerly a flourishing fishing port which now lies miles from the sea shore, is but one example. The net effect of all these abstractions is reduction in river flows, with resultant impact on in-stream biota and habitats.

Regarding indirect impacts, replacement of permeable, vegetated soil with impermeable surfaces of concrete and tarmac is the most extreme indirect change: in simple terms, perhaps only 5% of rainfall forms floods in a rural basin whereas 60% of rainfall may be converted into stormflow in an urbanized catchment. The implications are obvious: a much more flashy regime with greatly increased potential for erosion during flood events. This can destabilise in-stream and riparian habitats by creating a much more extreme flow regime, and also impact human dwellings near the banks during times of flood.

The influence of small changes, even if dramatically changing the local hydrological response (e.g. urbanisation), tends to be undetectable within the flow regime of a large river basin. However, where changes to land cover are sufficiently widespread, then regimes of even large rivers can be significantly altered, with implications for all those dependent on the river, including in-stream biota. Box 2.2 describes how hydrologists conduct "paired catchment experiments" to investigate the effect of land use change on the water balance. Figure 2.10 shows the difference in annual runoff for a treated catchment Watershed (WS)1 compared to its control, WS2. The control period ran from 1953 to 1962 and then the cover of mature Douglas fir trees was logged. From 1967, vegetation cover on WS1 has been allowed to re-grow naturally, but even after more than 40 years, it is clear that there is still more than 100 mm extra runoff each year from WS1 compared to WS2. This is because the vegetation cover on WS2 is old-growth forest: the canopy intercepts large quantities of rain and snow which is then evaporated without ever reaching the ground. The new trees, even when thirty or forty years old, still intercept less water than the fully mature tress, so it may be several decades more before there is no difference between

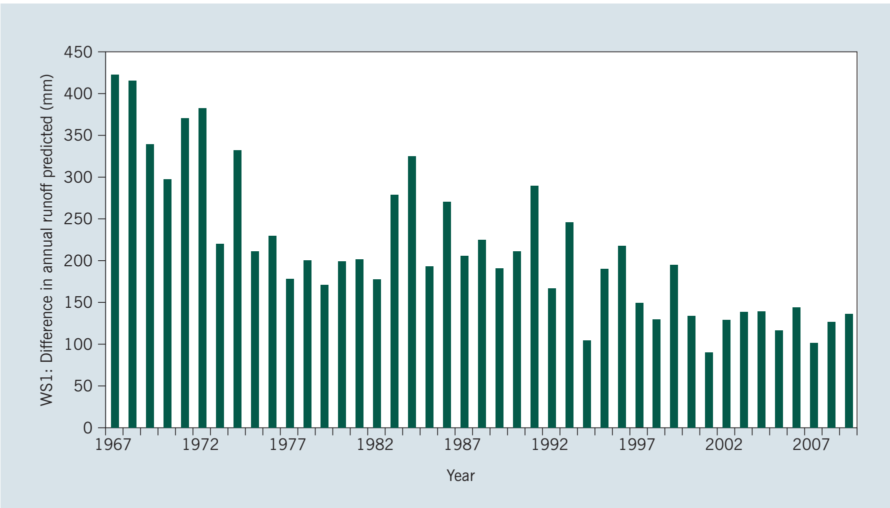


Figure 2.10: *Difference between predicted and actual annual water yield from WS1, H J Andrews Experimental Forest, Oregon*

Source: CLIMDB/HYDRODB database. The prediction method is explained in Box 2.2.

Box 2.2

Water or wood? Experimental investigations in forest hydrology

To investigate the impact of land use change on the water balance, the traditional approach in hydrology has been to conduct "paired catchment" experiments. Two adjacent basins, otherwise as similar as possible in all respects, are studied together. First, there is a period of calibration (ideally several years), during which a regression equation is established to show the relationship between runoff from the two basins. Then, land cover is changed in one basin ("treatment") whilst the other remains unchanged ("control"). The impact of the land use change is shown by calculating the difference between the actual response and that predicted using data from the control catchment.

Paired catchment studies in forest hydrology were pioneered in the USA. Figure 2.11 shows results from a series of treatments at the Coweeta Hydrologic Laboratory, North Carolina, USA (Burt and Swank 1992). Following clearance of the hardwood forest, to be replaced with grass, water yields increase, but the exact amount depends on the vigour of grass growth. When the grass is fertilized and dense, the overall effect is little different from the original forest, but as the grass productivity declines, because it was no longer receiving fertiliser, water yields rise. This is because there was a less dense leaf canopy so less rain was intercepted and lost through evaporation. When herbicide was applied to kill the grass, water yields increase dramatically, since there was no transpiration and the dead grass intercepts little of the rainfall. Finally, as the natural forest is allowed to regenerate naturally, water yields gradually decline towards the expected level. Such changes would, of course, impact on in-stream conditions, especially in headwater reaches where riparian shading is lost when the tress are cut down.

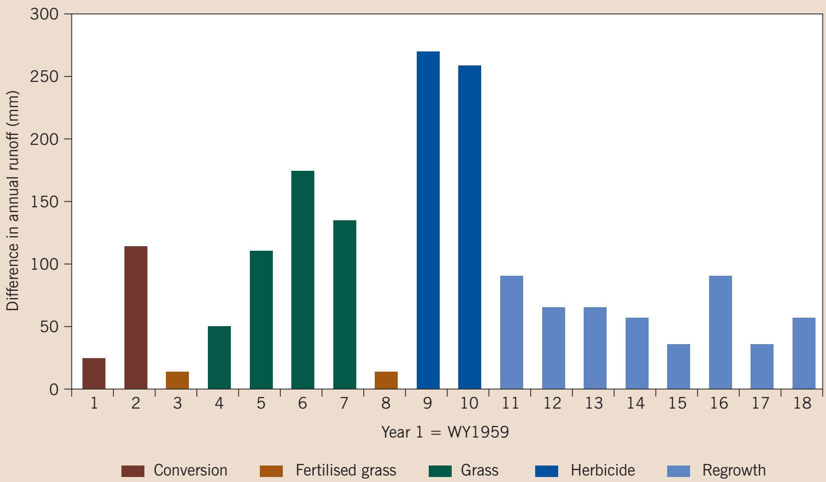


Figure 2.11: *Changes in annual water yield on WS6 at the Coweeta Hydrologic Laboratory, North Carolina, following forest clearing and application of different treatments. (See text for details)*

Source: Adapted from Burt and Swank (1992).

the water balance of the two basins. Thus, the impact of deforestation can be very long-lasting, even when the forest cover is allowed to recover. Of course, deforestation affects not just the amount of runoff: shading of the channel is lost, runoff pathways are significantly altered, and the river's sediment and solute load both change too. There are dramatic implications for the channel biota and their habitats therefore.

Goudie (2006) has reviewed the scale and extent of global environmental change. Some changes are *systemic*, affecting the whole world, such as the impact of global climate change resulting from emissions of greenhouse gases. Other changes are *cumulative*, indicating the substantial and significant accumulation of localized changes. It is these latter changes that characterize the human impact on rivers and river basins. The scale of cumulative land use change since pre-historic times is dramatic: Goudie (2006) shows that the area of forest has declined from 46.8 M km^2 to 39.3 M km^2 today, a reduction of 16%. Loss of grassland (19%) is larger in percentage terms but involves a smaller land area (from 34 M km^2 down to 27.4 M km^2). At the same time, the area of cultivation has increased from nothing to 17.6 M km^2 and the combined size of the global urban area is thought to be around 2 M km^2. As a result of all these various changes, relatively few large rivers can remain in pristine condition, and any assessment of future change must be judged against the uncertain baseline of "recent" condition.

2.8. Future projections

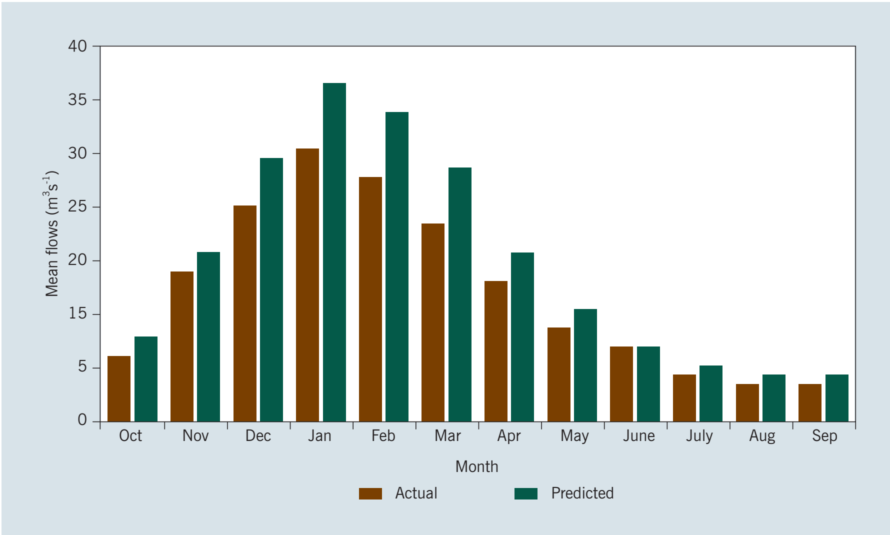
We have already noted that the main drivers of long-term variations in river flow are climate and human impact. What does the future hold for our river basins? It seems all too likely that people will continue to have significant impact on basin hydrology (Pearce 2007), especially in the developing world, as ever more food is needed for growing populations.

The main drivers of long-term changes in rover flow are climate change and human impact

Two examples will serve to illustrate these long-term changes. In Figure 2.12, a comparison is made between actual flows in the River Thames, England, using actual flow data from a gauging station in the middle of the basin, and a predicted flow series derived from rainfall data. Actual flows are significantly below predicted right across the flow range. For example, the mean summer flow (June – August inclusive) is actually 5.4 m^3s^{-1} but is 6.4 m^3s^{-1} for the reconstructed series. The mean monthly flow exceeded in 95% of months is 1.37 m^3s^{-1} for the actual series but is 2.18 m^3s^{-1} for the reconstructed series.

The second example of human impact comes from the Ebro River in north-east Spain. The Ebro River basin contains 187 dams, with a total capacity equivalent

Figure 2.12: *Mean monthly flows for the Thames at Eynsham (NRFA station number 39008) and for the reconstructed flow series*



Source: http://www.cru.uea.ac.uk/cru/data/riverflow/

to 57% of the total mean annual runoff (Batalla et al. 2004). The diverted water is used mainly for hydro-electric power production and for irrigation. Virtually all the dams were constructed during the 20th century, two thirds in the period 1950-1975. Figure 2.13 shows the highly significant decline in annual mean flow since 1954; the Tortosa gauging station is close to the Mediterranean Sea and therefore integrates the response for the entire basin. For the period 1917-1931 i.e. prior to the Spanish Civil War, the mean flow was 567 m^3s^{-1}. For the period 1954-1975, the mean flow was 493 m^3s^{-1} and only 309 m^3s^{-1} for the period 1976-2007. This is clear evidence therefore that the Ebro basin has been significantly impacted by the construction of reservoirs. It is worth adding that some head-water basins draining the southern Pyrenees have seen a considerable increase in forest cover due to both land abandonment and afforestation (Gallart and Llorens 2004). This will have contributed to the decline in flow as there are higher evaporation losses from forest than low crops (see also Box 2.2).

2.9. Is our glass half full or half empty?

The global population has now reached 7 billion, half of whom live in cities; the global population is expected to reach between 7.5 and 10.5 billion by 2050. Given an extra 0.5-3.5 billion mouths to feed, plus the rising expectations of a developing world, we can anticipate continued pressure on the world's resources, water most especially. At the same time, there is growing recognition of the

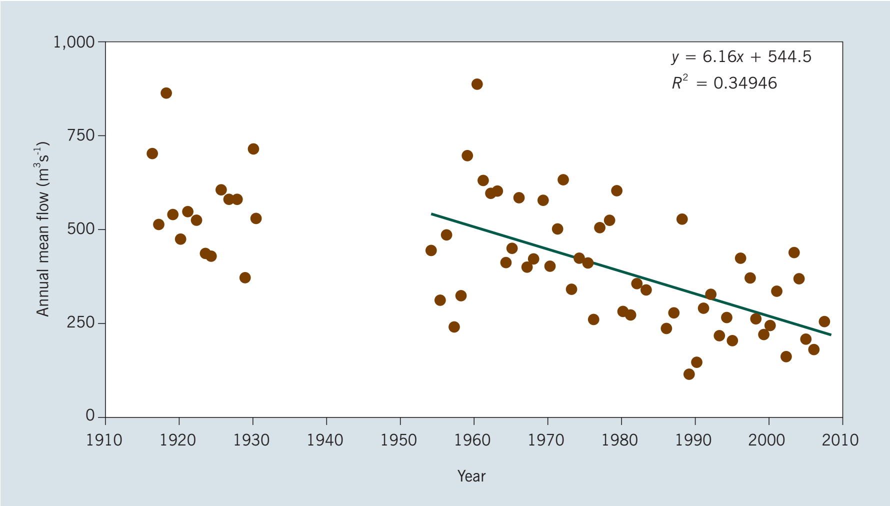


Figure 2.13: *Annual mean flow (m^3s^{-1}) at Tortosa, Ebro River, Spain (The base year was 1954)*

Source: Data from the River Ebro Authority (Confederación Hidrográfica del Ebro).

need to create sustainable water use, not threatening the opportunities for future generations by excessive exploitation today.

Should we be optimistic or pessimistic about the future of our rivers? In the near future, we can expect to see greater pressure on water resources in the developing world, with more dams being built, for example along the Mekong River, and more water being abstracted for irrigation. This will have serious consequences for river conservation. Meanwhile, in the developed world, there are likely to be efforts to reverse some of the worst effects of river engineering, with some dams being removed. Wiser use of water may even lead to a slight fall in consumer demand. Under these circumstances, we might speculate that climate will become a relatively more important driver of long-term change in river systems.

Gradually, we may see some rivers becoming noisy again, as the flow regime is returned to its natural state. This certainly means higher flood peaks. It may mean higher baseflow too, if water abstractions are reduced, but it might mean lower – but more natural – baseflow when dams are removed since "compensation water" is often released from reservoirs to maintain a more even flow regime. However, in most river basins, particularly in the arid regions as well as in the poorest countries, as we struggle to provide drinking water and to grow crops, rivers will become silent. Not only will there be deterioration of physical habitat; other problems will include decreased dilution capacity, reduced possibilities for fish migration and much less frequent flooding of riparian areas. Fred Pearce advocates a more sensitive use of water resources, giving up the notion of the "technological fix" and learning to treat nature as the ultimate provider of water, not a wasteful withholder. We cannot keep turning to irrigation when the climate gets drier or build more levees to prevent flooding in wetter periods. Treating water as a precious resource will allow us to better protect our rivers as well as deal with human needs. We need to give water back to nature, to protect our wetlands and conserve our freshwater ecosystems. This approach is reflected in modern legislation, such as the European *Water Framework Directive* (WFD: 2000/60/EC). With its holistic approach focusing on the achievement of "good ecological status", the WFD will help deliver water quality that favours the health of aquatic habitats as well as the quality of drinking water. Protecting and improving the riverine environment is an important part of achieving sustainable development and is vital for our long-term health, well-being and prosperity as well as for the river basin in which we live.

Grateful thanks to Chris Orton, Cartography Unit, Geography Department, Durham University, who drew the map and diagrams.

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Chapter 3

River's Architecture Supporting Life

Ramon J. Batalla and Damià Vericat

Physical and biological processes are inseparable in rivers. Fluvial ecosystems are adapted to change and need physical instability (floods) to keep ecological integrity. Impacts such as damming, channelization, and changes in land use, alter river dynamics. Rivers are distinct from each other, and so are solutions to their problems. Available scientific and technological knowledge must be the bases for a sound river management.

3.1. River form: The starting point for conservation

Rivers are among the most complex and dynamic systems in nature. They constitute natural units characterized by more or less frequent transfers of water and sediments that, in turn, support life. While moving through stream courses, water and sediments connect all river compartments, from the basin headwaters to the lowland deposition zones (e.g. Leopold et al. 1964; Richards 1983). The failure to appreciate this fundamental connection underlies many of the current environmental problems in river conservation and management.

River form and sediments create and maintain a variety of instream habitats that support the life of many organisms. A river habitat refers to the substrate, flowing water, organic debris, amongst others, which provide support for organisms i.e. animals and plants that live in the stream. Many river features can be distinguished, including *Riffles*, shallow zones where the water flows swiftly over rocks; *Pools*, deeper zones with more slow water; *Bars*, accumulations of sediments forming islands and forcing the water through secondary channels; and *Oxbows*, lakes formed when a meander is cut off. Each one of these habitats offers opportunities to different assemblages of organisms. Even within a single river habitat, there are many places where different animals can live. For instance, in a single riffle some organisms can select the areas with fastest current; some others seek shelter behind rocks. Riparian vegetation, the plants that grow in the floodable banks, is also an important element in the architecture of many rivers, and they exert a strong influence on river conditions (Chapter 9). Therefore, healthy riparian zones are key to a healthy in-stream habitat.

Physical and biological processes are inseparable in river systems; and they need geomorphic disturbance to keep ecological integrity

Humans modify and alter rivers. They can affect the physical functioning of fluvial systems both by land-use changes at the basin scale and by within-channel activities, singularly dams, channelization and gravel mining. All these perturbations alter water and sediment delivery to the drainage network, and the mass and energy transfer within it. Changes in land use (i.e. afforestation, deforestation, urbanization) affect runoff and sediment supply at the large scale and in the long term (Chapter 2). In turn, dams affect the water flow regime and sediment delivery over the long term and over long distances. Channelization, leveeing and rip-rapping transform channel geometry, change hydraulic properties of the flow and disconnect the streamcourse from its alluvial plain. Instream mining (i.e. extraction of sediments from streams and adjacent floodplains) acts locally by depleting the channel of sediments, and its effects can propagate down and upstream over decades.

Within this context, this chapter aims at providing a general view of the importance of the interaction between flow forces and sediments to shape rivers, and their relation with ecosystem functioning. We therefore introduce concepts of fluvial geomorphology and show selected examples to illustrate the discourse. Examples do not seek to be exhaustive and right away subject to extrapolation, but simply constitute a basis to interpret the geomorphic (i.e. physical) contribution to river ecological integrity. Analysis of physical processes provides a comprehensive framework for river sciences, enabling us to view water, sediments, and resultant physical features as fundamental elements to understand and inform conservation and restoration measures in the system. Conservation and restoration starts from the understanding of river physical processes and dynamics. River management that neglects focussing on mass and energy balances as the factors driving river functioning is bound to failure.

3.2. What is in there? Water, pebbles… and sometimes mud

Sediments of different sizes and shapes form the complex architecture of streamcourses. Heterogeneity of substrate guarantees the maximum ecological diversity in a fluvial system

The structure of alluvial river channels (namely, its basic *architecture*) is formed of sediments that experience cycles of entrainment, transport and deposition (e.g. Church 2006). Most rivers on Earth are alluvial, i.e., water runs through loose mixtures of sediments that have been previously deposited. These sediments form the basic structure of rivers; within them water moves upwards and downwards, and laterally in direct connection with groundwater in the floodplains; sediments host a variety of fauna and flora, and support riparian vegetation.

The architecture of river channels is controlled by the interactions between water discharge, the size and sorting of bed sediments, the supply of new sediments from the catchment, and their transport downstream. These interactions control the moments in which river channels change and their temporal sequence and magnitude. Floods are physical disturbances for river-dwelling organisms.

From the point of river morphology, fluvial sediments can be divided into bed material and wash material. Bed material corresponds to the coarse sediments supporting the channel and banks and, ultimately, determines the form of streamcourses (Figure 3.1A and 3.1B), which is an important part of river habitats.

On the other hand, wash material correspond to fine sediments transported for longer distances in suspension. Wash material does not determine the form of alluvial channels but influences the upper bank morphology (Church 2006). These fine sediments are often deposited into the coarser bed material, clogging the near-surface pores and thus affecting the structure of the framework (Figure 3.1C). This effect may influence the cohesiveness of bed sediments and their stability and alter habitat conditions for biota, for instance, by reducing refuge in the interstitial space. Hence, physical characteristics are key factors controlling habitat conditions that are essential to maintaining the ecological diversity of a particular fluvial system.

Ecological diversity of river ecosystems is directly linked to the heterogeneity of physical habitat conditions, including flow hydraulics and substrate. However, there are important scale considerations related to organism size. For invertebrates the relevant scale of physical heterogeneity is that of the patch (i.e. centimetres to metres), whereas fish, which are larger and more mobile, depend on heterogeneity at the reach-scale, hundreds of metres to kilometres (Poff 1997). This functional relation between spatial scale and optimal ecological diversity is

Figure 3.1: *A) A mountain reach with a complex architecture including riffles, pools, central bars and secondary channels (Feshie River, Scotland, UK; arrow shows flow direction). The inset zoom shows how morphological complexity changes in relation to the scale in which it is investigated. B) Sediments are mixed horizontally and vertically: Gravel and cobble sediments in a complex arrangement (Ribera Salada, Southern Pyrenees). C) Fine sediments deposited during low flows clog the spaces between gravel particles (Isábena River, Southern Pyrenees)*

conceptually represented in Figure 3.2. Sediments of different sizes and shapes form the complex architecture of streamcourses (i.e. their form). Substrate heterogeneity and topographic complexity are requisites for ecological diversity, which in turn is linked to river health.

3.3. Shaking beds move organisms: Life requires complexity and change

Relatively immobile bed sediments (i.e. cobble-boulders) are important habitats for invertebrates. These large particles offer a more diverse habitat for colonization and better food resources (i.e. because of organic material that they can retain or the more developed biofilm that can grow on them) than less stable environments. This contrasts with the more mobile sand and gravel, where even small increases in flow move particles and scour benthic animals. Thus, the reach-scale habitat diversity depends on the relative availability of stable and unstable areas of stream bed, as well as refugia and still waters. In rivers with high contents of fine sediments, siltation blocks the transport of oxygenated water to the sediments and thus results in death salmon eggs and other fish. Besides, clogging of beds by fine sediments also reduces invertebrate diversity

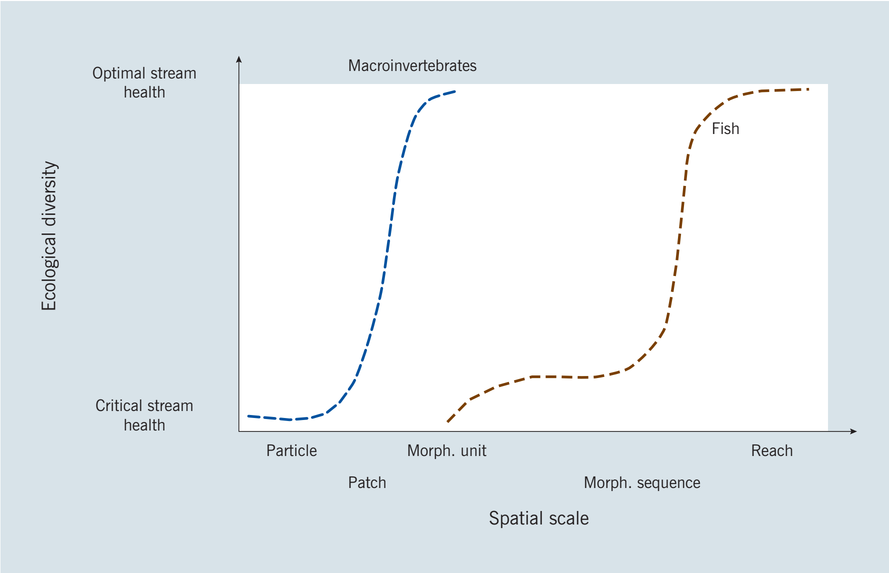


Figure 3.2: *Functional relations between spatial scale and optimal ecological diversity. Ecological diversity is directly linked to the heterogeneity of physical habitat conditions, including flow hydraulics and substrate, but invertebrates respond at smaller spatial scales than fishes (Note that the term* Morphological unit *refers to single elements present in a river channel i.e. bar, riffle, pool; whereas* Morphological sequence *refers to groups of units that alternate in the river channel i.e. riffle-pool sequence)*

(Gibbins et al. 2007). In any case, river beds experience disturbance (floods) from time to time. Flood frequency and magnitude depend on climate and basin characteristics (Chapter 2), but their disturbance effects are relatively larger in reaches where sediments move more easily. Thus, hydrology and sedimentology interact to control habitat diversity and functionality.

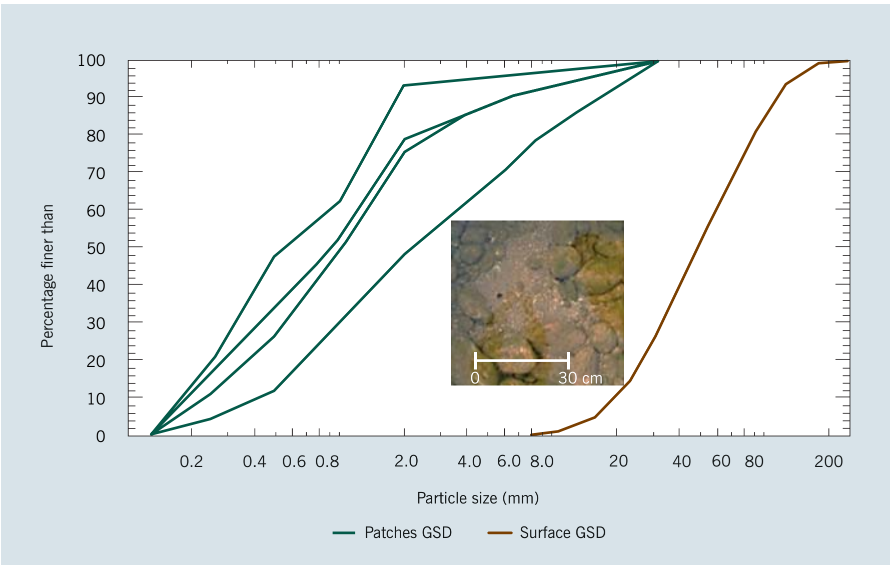
Channel shape and sediment size are related to flow energy, expressed by the combination of discharge and gradient of the river. This relationship commonly referred to as Lane's Balance shows that a change in any of the variables will cause a change in the others such that equilibrium is restored. When a channel is in equilibrium the sediment being transported into the reach is transported out of it, without significant deposition of sediment in the bed (aggradation, or building up of sediments), or excessive bed scour (degradation, or downcutting of the channel). It should be noted that by this definition of stability, a channel is free to migrate laterally by eroding one of its banks and building sediments on the one opposite at a similar rate. When the supply of water or sediments are changed channel geometry and bed composition adjust towards new configurations. These changes can result from many different causes, from changes in erosion rates in the basin to changes in climate. Changes in channel geometry also occur as the discharge rises and falls during the year, but these changes are frequently minor.

Floods determine the disturbance regime (i.e. frequency and magnitude) experienced by a given reach and, consequently, the associated ecological responses.

Hydrological variability is considered the main factor affecting the organization of riverine communities, contributes to key ecological processes (Yount and Niemi 1990), and is essential for river conservation and renaturalization. The temporal persistence of invertebrates or fish communities is determined not only by the resistance of the communities, but also by their rate of recovery from a given perturbation (Poff et al. 1997). When a flood occurs, flow energy dissipates along the streamcourses, eventually eroding channel bottom and banks, thus temporarily altering the *normal* (i.e. usual) habitat conditions. But even with small increments in river discharge (i.e. well below flood episodes), parts of the bed may get disturbed and community alterations occur. This is the case, for instance, of invertebrate communities living in patches of fine sediment usually located behind obstacles or in depressions in the river-bed (Laronne et al. 2001, Figure 3.3). Patches constitute an excellent example of the bio-physical complexity of rivers. Patches of fine sediment are the first to be moved when the flow rises, and their scouring can trigger massive invertebrate drift, and therefore facilitate passive downstream movement of those individuals.

River science has often faced difficulties in matching the study of physical and biological elements; this fact may be due to diverging objectives of the scientists analysing one or the other element, but also to technical limitations on sampling and modelling. Field experiments are not easy to carry out but they may

Figure 3.3: *Typical grain-size distribution (GSD) of patches (the fine particles in the inset photo and the coarse surface material (the larger particles in the inset photo). Example from the gravel-bedded Ribera Salada (Southern Pyrenees)*



Source: Redrawn from Gibbins et al. (2007b).

shed some light onto this type of interactive processes. As an example, Vericat et al. (2007) developed a portable flume (Figures 3.4A and 3.4B) that can be placed *in situ* within riverchannels and be used to modify local water velocity.[1] This field experimentation has shown that a small amount of bedload transport suffices to trigger massive invertebrate drift, demonstrating that magnitudes of physical and biological disturbance are often out of phase.

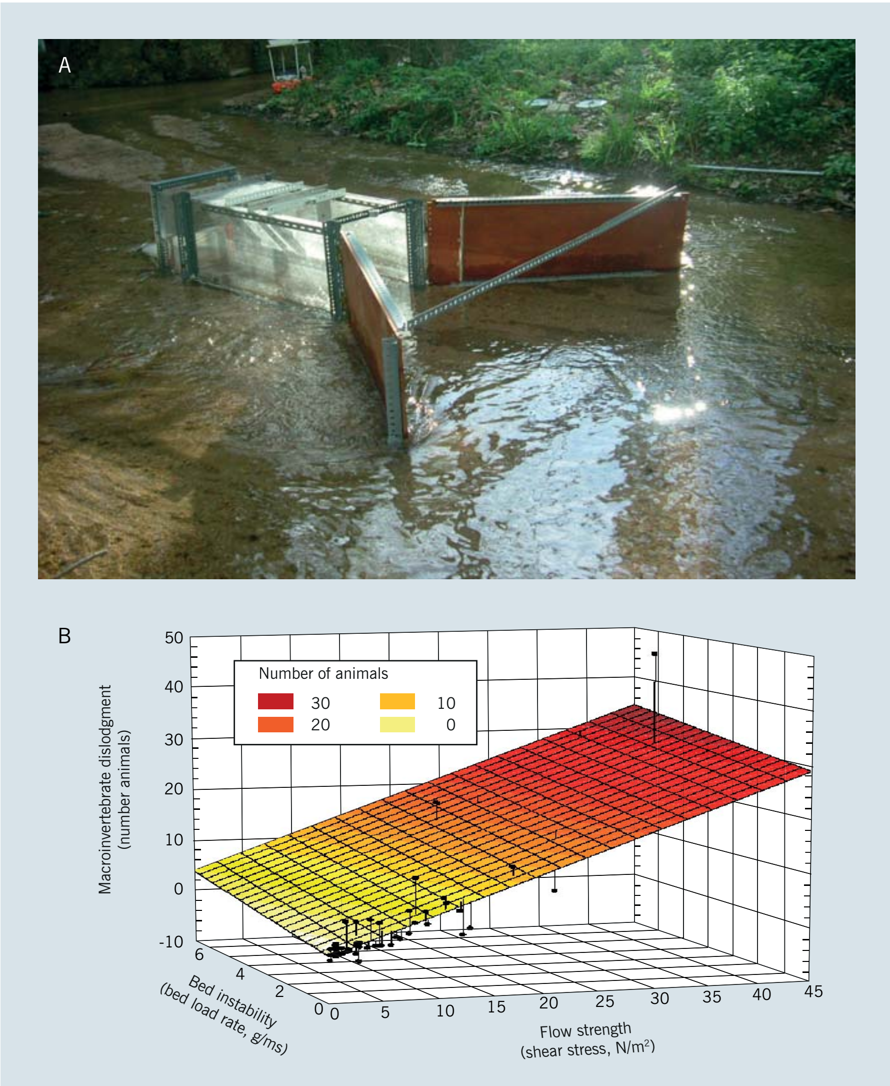


Figure 3.4:
A) Portable flume (Vericat et al. 2007) used to manipulate hydraulic conditions over patches of stream bed. It helps study the interactions between hydraulics, sediment transport and invertebrate drift. The wooden doors used to manipulate hydraulic conditions inside the flume are shown here in their open position, i.e. forcing the water velocity to increase. B) Diagram showing a 3-D model of the relations between flow strength (shear stress), bed mobility (bed load transport rate) and the loss of animals from patches of the gravel bedded Ribera Salada. Black points represent the raw data values, while coloured areas the modelled values. For more details on those biophysical relations see Gibbins et al. (2007)

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[1] See http://www.agu.org/pubs/eos-news/supplements/2007/41-410.shtml for details.

3.4. Rivers react to human actions

We have so far examined the effect of natural perturbations; but today it is particularly important to understand how humans interfere with natural processes, and how natural processes may be preserved and/or restored. From the many types of impacts on river channels and their basins, some act locally and have short duration, while others propagate over longer terms and distances (see Table 3.1). We will focus on two common disturbances affecting physical processes: dams (long-term) and gravel mining (local, short-term).

Rivers have been the main water resource for humans over history. Economic development following industrial revolution in many countries, singularly in Europe, was linked to increased demand for water and energy. Rivers supplied both. Many streamcourses have been progressively dammed through human history. Particularly, large rivers started to be regulated mostly since the 19th century. Regulation has grown exponentially through the 20th century and has permitted increasing water supply and hydropower production, well beyond the intrinsic climatic variability of many regions. Worldwide there are more than 45,000 large dams (larger than 3×10^6 m^3).[2] Arid regions account for the highest number of reservoirs/dams. A paradigm is the Iberian Peninsula, a semi-arid and water thirsty country, which assembles approximately 3% of the world's dams, mostly dedicated to agricultural, industrial and urban demand. The effects of reservoirs on flow regime depend on their size relative to river runoff, their purpose (e.g. irrigation, hydropower, flood control), and their operating rules. This complexity precludes simple generalisations about the effect of dams on discharge distribution (Williams and Wolman 1984).

Table 3.1: *Main human impacts affecting water and sediment-related processes in rivers, and their extension over different space and time scales (see Figure 3.5 for illustrations)*

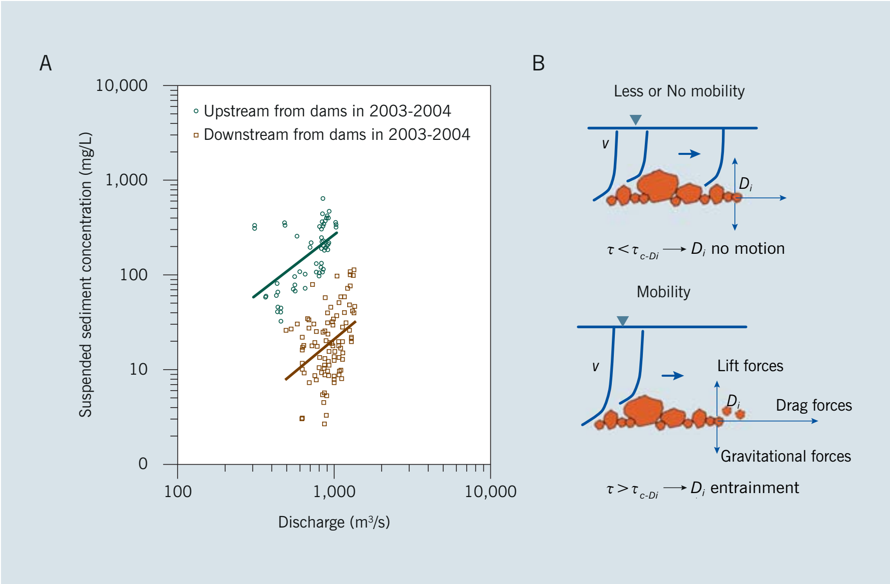
	Local ⟶		General
	Riverchannel	***Floodplain***	*Basin*
Short-term *from year to decade* ↓	• Gravel mining • Rip-rapping • Channelization	• Gravel mining • Channelization	• Land use changes (i.e. forest fires, urbanization)
Long-term *… decades to centuries*	• Gravel Mining • Rip-rapping • Channelization	• Gravel Mining • Rip-rapping • Channelization	• Dams • Land use changes (i.e. afforestation, deforestation)

[2] See International Commission of Large Dams at http://www.icold-cigb.net for more details.

Figure 3.5:
Examples of morphological impacts on rivers. A1) Incision and A2) disruption of channel form in the Ribera Salada (Southern Pyrenees) as a consequence of gravel mining. B1) Alteration of river continuity by damming, and B2) sediments retained in Barasona Reservoir, Ésera River, Southern Pyrenees). C) Rip-rapping in the Ribera Salada (Southern Pyrenees)

Overall, dams reduce flood magnitude and frequency (Batalla et al. 2004) and block sediment transport (Figure 3.6A) (Vericat and Batalla 2006), altogether reducing flow energy and sediment mobility (Figure 3.6B). The effects of dams are relatively larger on rivers in dry climates, both through reductions in high flows (reduced disturbance) and extended baseflows, making these environments more suitable for exotic species not adapted to seasonal drought (e.g. Batalla and Vericat 2009). Sediment transfer to downstream reaches is also altered. Virtually all bedload, and much of the suspended load are trapped into reservoirs. This sediment deficit generates a series of impacts on channel morphology and sediment characteristics. Loss of bars and other areas with bare sediments, intrusion

Figure 3.6:
A) Suspended sediment transport upstream and downstream from the dams in the lower Ebro River (data from Vericat and Batalla 2006). B) Conceptual model of bed mobility. Bed material entrains when the flow exceeds the critical strength for mobility. In the case of river channels downstream from dams, if flood magnitude is reduced, energy expenditure is less over river bottom sediments, hence reducing bed mobility and, with it, the natural perturbations basic to maintaining fluvial ecosystem functioning



Note: v = flow velocity, D_i = particle of an i diameter, τ = shear stress, $t_{c\text{–}Di}$ = critical shear stress for a given particle size i.

of terrestrial vegetation in formerly open areas (Williams and Wolman 1984, Figure 3.7), channel narrowing and associated changes in river flow conditions, are amongst the most pronounced physical effects downstream from dams. Water released by dams is often called *hungry water* (as per Kondolf 1997), as it leaves the reservoir with almost no sediment, and so, erodes sediments from the river bed without replacing them with new sediments from upstream. This fact creates a disequilibrium that may produce armouring of the river bed (i.e. only the largest particles stay in place; Williams and Wolman 1984) and incision of the channel (i.e. deepening, Kondolf 1997). These changes in flow and flood regimes and in channel form and sediments have important effects on the river ecosystem (Ligon et al. 1995). The modified regime exacerbates species with life history characteristics atypical of the pre-dam environment, including non-native species, resulting in altered species composition and vegetation dynamics (Cowell and Dyer 2002).

River sediments are naturally sorted and often close to markets, and thus, they have been widely used as a source of construction materials. Sediment mining affects streams and floodplains and it is severe in countries subjected to a rapid urban growth, where the availability of aggregate (sediment mixtures i.e. sand, gravels, used for construction) is key to maintaining economic activity (Kondolf 1997). Additionally, sediments are also extracted from highly dynamic rivers where sediments tend to accumulate in the channel, with the aim of

Figure 3.7:
Channel narrowing and vegetation encroachment as a consequence of dams in Segre River near the Alcarràs (Ebro basin, NE Spain). A) Segre River, 1956; B) Segre River, 2009

maintaining flood capacity (e.g. the Lower Waimakariri, New Zealand, Griffiths 1979). In other places (e.g. the River Platte, USA) sediments have been removed from islands to improve bird nesting habitat (Kinzel 2009). Sediment mining represents a non-natural stressor which profoundly modifies physical and ecological processes and dynamics. In contrast to dams, whose effects extend progressively over space and time, mining is a localized intensive impact. Once mining ceases, recovery of ecological diversity may require more time than after natural perturbations, even large catastrophic floods. The recovery time depends on the channel condition (physical and biological) after the impact, the flood and sediment transport regimes, including sediment availability and supply, and the distribution and dispersal ability of potential colonists.

Thus, both physical and ecological impacts of sediment mining leave short- and long-term signatures (Erskine 1997). Short-term impacts are those related directly to the mining activity, such as a turbidity plume or water barriers to fish migration. Long-term morphological effects include channel deepening and instability, and coarsening of the riverbed surface. Ecological effects including habitat homogenisation (e.g. Wyzga et al. 2001), result in decreased diversity and changes in species composition of invertebrates, biofilm and fish communities (Brown et al. 1998).

3.5. Floods: When the water dances with sediments. Opportunities for restoration

Floods are the most common form of natural disturbance in rivers. They constitute an essential element of the fluvial dynamics and, although sometimes may be the cause of economic damages, they are indispensable for the river's normal functioning (Chapter 2). Dams are the elements that most directly alter the flow regime, mostly by absorbing flood flows and collecting almost all the sediment carried down in the river basin. Floods and sediments are key elements for the good functioning of river ecosystems.

Sound management of available water in the catchment may return a certain degree of naturalness to a river. Conserving and/or restoring the natural variability of the river flow is a worthwhile way to progress towards that goal. In particular, artificial flow releases from dams, known as *flushing flows*, provide an interesting opportunity to restore river processes in altered streamchannels. They can be designed to modify or maintain the channel sediment and geometry (Kondolf and Wilcock 1996) or the riverine ecosystem as a whole (Arthington and Pusey 2003). Milhous (1990) provided some rules to estimate the flushing flow needed to keep the substrate in a condition that will support a desired aquatic ecosystem. For instance, and in order to remove interstitial fine sediment from gravels, we can calculate, based on the median size of the gravel, the critical shear stress necessary to set gravels in motion; once gravel particles are entrained into motion, sand beneath them may be entrained and removed from the bed. However, the use of hydro-geomorphological criteria both fixed (i.e. river-bed grain-size distribution) and dynamic (i.e. sediment transport) is still not very common (e.g. Kondolf and Wilcock 1996, Batalla and Vericat 2009). Despite several constraints, if carefully designed and implemented, flushing flows may play an important role in enhancing physical habitat in the river. Flushing flows can also be suitable in rivers affected by hydropower production, and may actually result in a positive trade-off due to vegetation removal and reduced clogging of water intakes (Batalla and Vericat 2009). It is, however, necessary to reassess their effectiveness regularly and monitor adverse physical effects like riverbed erosion. Flushing flows are an important instrument of river management, but one that must be employed as part of a spectrum of approaches to enhance physical habitat conditions and restore basic river functions.

Riverchannel instability maintains streams alive and must form the core of conservation and restoration practices

Complementarily, sediment extracted from reservoirs or debris-control basins has been utilized to enhance fish habitats. This practice is known as *gravel replenishment* and has been implemented in Sacramento River, California, downstream from the Keswick dam (Buer 1994). This type of actions provide short-term habitat, since the amount of gravel added is but a small fraction of the bedload deficit, and gravels placed in the main river can be typically washed out during high flows, requiring continued addition of more gravel (Kondolf 1997). In the Rhine River sediment injection has been implemented downstream of the Iffezheim dam. This approach has proved successful in preventing further incision of the riverbed downstream and to protect river infrastructure (Kuhl 1992).

3.6. Maintaining river form and processes: A way to keep rivers active

Most rivers are not and will no longer be pristine anymore. All societal bodies (i.e. authorities, scientists, environmentalists, company managers and, overall, citizens as end-users) must accept and agree on this fact. The question arises of how to make compatible the use of natural resources (surface waters, in this case) and the conservation of river integrity as its most important element. Recipes are not universal and must be kept simple to guarantee probabilities of success. A few final remarks and recommendations encompassing the main concepts outlined in this chapter can be drawn as follows:

— Physical and biological processes in rivers must be seen as inseparable. Water and sediment dynamism constitute the bases to maintain the ecological integrity of a river system.
— Rivers need to maintain physical disturbance (i.e. floods). Physical instability keeps streamcourses active and must form the core of conservation and restoration plans, if accompanied by evaluation programmes based on monitoring, sampling and modelling.
— Available scientific and technical expertise is already sufficient and ready to inform river management practices. Continuous reassessment of renaturalization and restoration practices is a key factor to keep work in progress and updated. Twenty-first century technical developments support the implementation of sound guidelines to the fields of river science and engineering.

In spite of being governed by universal factors, rivers are complex and distinct, and no universal solutions exist to face environmental problems in the whole variety of contrasted socioeconomic and climatic environments on Earth. Indeed, extrapolation between river basins is a smart way to progress, but local *ad hoc* actions (both short and long-term) such as, (i) flushing flows, (ii) sediment injection downstream from dams, (iii) sediment pass-through reservoirs, (iv) periodical reservoir drawdown and sediment dredging, (v) restoring of abandoned channels, (vi) decommissioning levees and re-introducing sediments into streams, among others, shall be put on the agenda and progressively implemented.

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CHAPTER 4

Nutrient Pollution: A Problem with Solutions

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Nutrient pollution of rivers is one of the most widespread human impacts on water resources. Wastewaters from urban and agricultural activities are the source of most nutrients, which stimulate excessive growths of algae. Algal blooms can physically alter the structure of habitats, increase productivity of food webs, decrease oxygen concentration, and increase pH of waters, which causes complex effects on the productivity and biodiversity of algae, invertebrates, and fish. At low and intermediate levels of nutrient pollution, productivity of invertebrates and fish can increase with nutrient pollution, but high levels of nutrient pollution cause low oxygen that reduces animal productivity. Whereas the number of all species of algae, invertebrates, and fish may not be reduced greatly by low and intermediate levels, the numbers of sensitive species are reduced. In addition to nutrient effects on biodiversity, nutrient pollution reduces the drinking water, recreational, and fisheries uses of rivers as well as the downstream receiving waters. Algae growing in high nutrient conditions commonly produce toxins that affect drinking water as well as aquatic biodiversity. Reductions in water transparency from algae and excessive growth of algae and aquatic plants on river bottoms can reduce value of rivers for boating, swimming, and fishing. Nutrients in rivers are transported to downstream lakes and coastal zones, where problems with hypoxia and harmful algal blooms are increasing around the world. Now is the time for developing comprehensive nutrient management strategies for rivers and downstream waters. Scientific evidence clearly shows that nutrients in rivers cause important problems that severely affect ecosystem services and human well being. Threshold responses by rivers to nutrient pollution help develop stakeholder consensus for management goals. Freshwater science is sufficient for developing site-specific management goals accounting for differences in uses of rivers, in river responses to nutrient pollution, and for regional needs. Cost effective strategies exist for reducing nutrient pollution. Scientists, policy makers, and other stakeholders should seize the opportunity to advance nutrient management in rivers and thereby improve and protect the ecosystems services provided by rivers and downstream waters.

4.1. Nutrients: Necessary but spelling of harmful when in excess

Nutrients are chemicals needed by organisms to survive, grow, and reproduce. Autotrophs are organisms needing only inorganic nutrients, such as water, carbon dioxide, nitrate, and phosphate, plus the energy from sunlight and photosynthesis to make the organic molecules that compose cell parts and enable growth and reproduction. Algae and aquatic plants are autotrophs in rivers. In contrast to autotrophs, heterotrophs need organic molecules for energy and for nutrition. Fungi and most bacteria, other than cyanobacteria, are heterotrophs that require organic molecules as a source of energy and a wide diversity of inorganic and organic chemicals for nutrition. The combination of chemicals needed by these microbes depends upon the species. Animals require organic molecules as a source of energy and nutrition. Thus, the basic supply of inorganic nutrients and sunlight regulate how rapidly organisms grow in an ecosystem, and often the biomass of organisms that occur. In river ecology and management, nutrients usually refer to the inorganic chemicals needed by autotrophs.

Inorganic nutrients occur naturally in ecosystems, originating from dissolution of rocks, the bacterial process of nitrogen fixation in which atmospheric nitrogen (N_2) is converted to ammonia (NH_3), and from decomposition of dead organisms by bacteria and fungi. Nutrients are transported to rivers via runoff and subsurface groundwater flows. Because types of rocks, terrestrial vegetation sequestering nutrients, and precipitation vary from one region to another, naturally occurring nutrient concentrations vary among rivers in regions with different geology and climate (Smith et al. 2003). Nutrient generating processes are usually relatively low compared to demand in ecosystems, so most ecosystems without nutrient pollution by humans have very low nutrient concentrations. The macronutrients nitrogen and phosphorus are important among all the nutrients in aquatic ecosystems, because they are usually in shortest supply compared to the others. When they are in short supply, they limit the rate that algae and plants can grow. Phosphate, nitrate, and ammonia are the forms of phosphorus and nitrogen used by algae and plants. In general, terrestrial and marine ecosystems tend to be more limited by nitrogen than phosphorus, and freshwater ecosystems tend to be more limited by phosphorus than nitrogen.

The sources of nutrients and impacts of nutrients on rivers and downstream waters are widespread (Carpenter et al. 1998; Smith 2003; Foley et al. 2005). Even in the US, with relatively low impacts to river catchments, almost half of the length of streams and rivers have been altered by nutrients. Nutrient alterations of ecosystems tend to be greatest in climatic and geological regions in which humans can develop cities and grow food, what Ellis and Ramankutty (2008) have called “anthropogenic biomes”. Most nutrient pollution originates from excess fertilization of terrestrial habitats (particularly croplands) and the waste of human and animal symbionts (chickens, cattle, pigs, etc.).

Nutrient pollution causes excessive growth of algae and plants, which leads to other imbalances in aquatic ecosystems. Excess algae and plant growth in aquatic habitats can: 1) physically alter habitats by overgrowing rocks, sands, and bottom sediments and 2) chemically alter habitats by reducing dissolved oxygen, increasing pH, and even producing toxins. Most aquatic species cannot tolerate low dissolved oxygen, high pH, and physically congested habitats. In addition, many naturally occurring species are adapted to living in low nutrient habitats. High nutrient concentrations allow invasion of species that require the higher levels of nutrients and productivity to survive, which can cause shifts in competitive balances and loss of species adapted to low nutrients and productivity. In addition to problems with nutrients altering biodiversity, the algae growing in high nutrient environments can produce toxins and precursors for toxins that foul drinking water, potentially increase persistence of pathogenic bacteria, and reduce aesthetic appeal of rivers as algae overgrow substrata and cloud the water. In both developed and undeveloped regions of the world, including many areas of Europe and the US, groundwater is contaminated with sufficiently high concentrations of nitrate that it is dangerous for human consumption (Townsend et al. 2003). “Blue-baby” syndrome (methemoglobinemia) and a diversity of cancers have been associated with high nitrate in drinking water.

Nutrient pollution of rivers also affects lakes and coastal zones. Nutrient pollution causes widespread problems with loss of biodiversity, drinking water, and recreational uses of water

Nutrient alteration of rivers also causes downstream impacts on lakes, estuaries, and coastal zones. Lewis (2011) estimates a 74 percent increase in algal and aquatic plant production in lakes since 1970. Seitzinger et al. (2010) estimated nutrient exports from rivers to coastal zones have increased 15 percent since 1970. The result has been extensive development of harmful algal blooms and low oxygen conditions in coastal zones around the world (Rabalais et al. 2010). Climate change as a result of global warming is expected to increase intensity of rainfall and flooding, which will increase nutrient transport from land and rivers to downstream waters. In addition, use of fertilizers and intensity of agriculture is expected to increase in the next 50 years as demand for food increases by a growing world population. So need for nutrient management in rivers is critical for both instream and downstream conditions.

The problems with managing nutrient pollution are somewhat different than other contaminants of rivers and other aquatic habitats. As with other

Box 4.1

Algae everywhere

Algae and aquatic plants are a highly diverse group of photosynthetic organisms that live in all aquatic habitats. Algae are distinguished from plants because they do not have sterile cells around reproductive structures. Since algal reproductive structures are sensitive to dry conditions, algae are restricted to life in water. Cyanobacteria were the first organisms that evolved the photosynthetic processes that produce oxygen, resulting in increased oxygen in the atmosphere of the earth over 2.5 billion years ago. Cyanobacteria, green algae, and diatoms are the three most common algae in most freshwater ecosystems. Green algae are green because they have a dominance of chlorophyll pigments in chloroplasts,

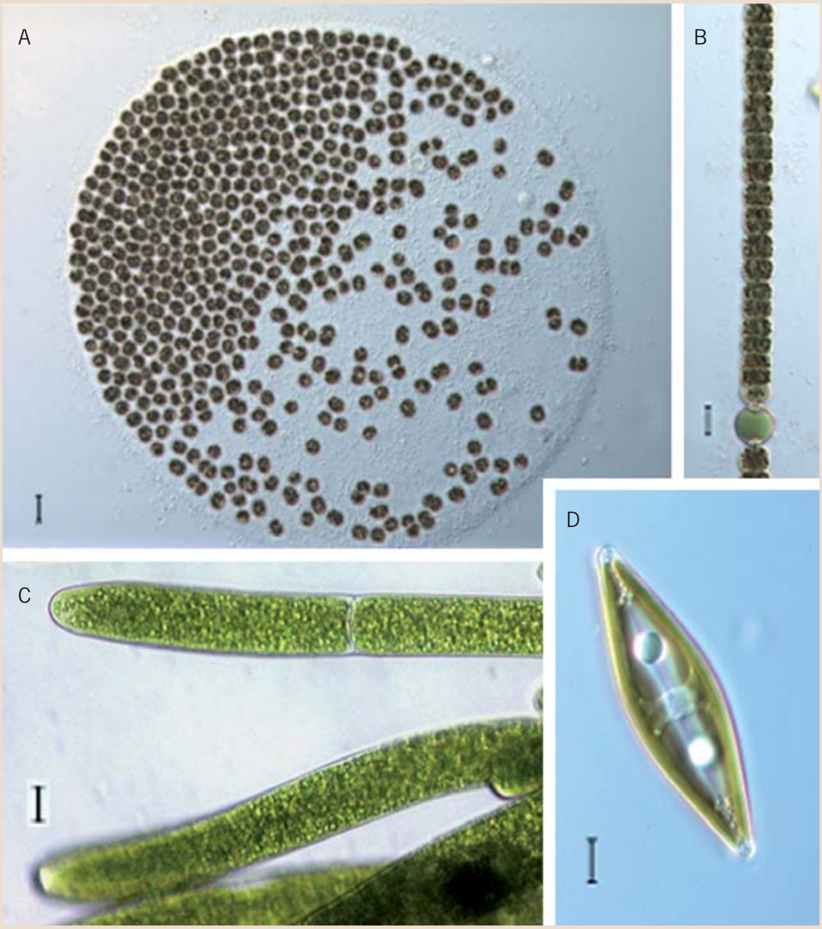


Figure 4.1: *A)* Microcystis, *a colonial cyanobacterium, which is known to produce toxins. B)* Anabaena, *a filamentous cyanobacterium with a heterocyst to fix nitrogen. C)* Cladophora, *a green algae. D)* Craticula, *a diatom. The scale bars in A-D indicate 10 µm*

which reflect green light. Although all three groups have green chlorophyll pigments, accessory pigments cause cyanobacteria to be blue-green and diatoms to be golden-brown. Cyanobacteria are unusual because they can fix atmospheric nitrogen into ammonia. Green algae have thick cellulose walls around each cell and store starch from excess photosynthesis. Diatoms have glass cell walls and store oil from excess photosynthesis. The glass cell wall is composed of two halves that separate during cell reproduction. Because of the glass cell wall, diatom growth can be limited by silica availability, as well as phosphorus and nitrogen availability. Aquatic plants range taxonomically from the primitive mosses that are common in headwater streams to the flowering plants. Some aquatic plants are adapted for fast current with long narrow leaves, whereas others may have floating leaves and live in margins of wetland streams and rivers.

contaminants, we are concerned about instream and downstream effects and the concentrations of contaminants that have negative effects on valued ecological attributes. Nutrients do not usually have direct toxic effects on organisms, so perceived risks by the public for nutrient contamination are not as great as contaminating valuable resources with toxic substances, like mercury and PCBs. However, scientific evidence is clear that high levels of nutrient pollution impair drinking water quality, public health, recreational uses of water, and biodiversity (Townsend et al. 2003; Downing et al. 2001; Suplee et al. 2008). Intermediate levels of nutrient pollution are not known to have great effects on drinking water quality and human health, but they can impair biodiversity. For some uses of rivers as well as the surrounding catchment, intermediate levels of nutrient pollution resulting from exploiting services of agricultural ecosystems can actually have positive effects on some ecosystem processes and some measures of biodiversity when high nutrient taxa invade. Effects of nutrients vary depending upon climatic and geological setting. Thus tradeoffs in managing rivers for one use or another and natural variability among regions present challenges for resource managers determining goals for resource management and pollution allowances that protect those goals.

In the following sections, we discuss effects of nutrients on biodiversity and human uses of rivers. We explore the effects of nutrients on algae, invertebrates, and fish as well as sources of nutrients. The challenges of measuring biodiversity and characterizing effects of nutrients on biodiversity are discussed. Finally, we discuss the possible solutions for land and waste management that can minimize nutrient pollution as well as strategies for reducing tradeoffs in managing rivers for their many uses.

4.2. Nutrient effects on algae

Nutrients enable growth of algae, plants, and bacteria in streams. Nutrient uptake rates, growth, and biomass accumulation rates increase asymptotically with increasing nutrient concentrations (Figure 4.2). Uptake occurs by active transport of nutrient ions through uptake sites in cell membranes, so uptake increases with nutrient concentration until all uptake sites are active. Nutrient uptake rates can exceed diffusion rates of nutrients to cells. In addition, as algae accumulate on substrata, flow of stream water through microscopic spaces among the algae slows. As a result, nutrient uptake and cell growth rates decrease with increasing algal density (Figure 4.2).

Algae-nutrient relationships become more complicated when put in the context of the complexity of river ecosystems. First, algae-nutrient relationships vary depending upon where algae are in the river. We should distinguish between benthic algae that are attached to the bottom of rivers and planktonic algae that are suspended in the water. Benthic and planktonic algae grow independently in their respective habitats, but they also interact as planktonic algae settle onto the bottom of rivers and grow and benthic algae drift from the river bottom and become suspended in the water column. With more light reaching the bottom of shallow streams, headwater and mid-sized rivers often have more benthic than planktonic algae. As waters flow slowly downstream planktonic algae grow and accumulate in the water column, reducing light penetration to the river bottom, and causing a shift in relative importance of planktonic algae over benthic algae in larger rivers. So nutrients generate problems with benthic algae in shallow streams and smaller rivers and planktonic algae in large rivers.

Nutrient effects on algae also vary depending upon the type of substratum in the river. Benthic algae can accumulate to much greater abundances when substrata are large cobble or bedrock that move relatively little in streams, because filamentous green macroalgae are more likely to grow in abundance on these substrata. Microalgae are the most common algae on smaller substrata and plants can grow in sediments.

Rain and resulting runoff to rivers and high flows can reset river ecosystems by scouring benthic algae from the bottom, washing planktonic algae to downstream lakes or the coastal zone, and replenishing nutrient supplies that may have been depleted during prior algae accumulation periods in the river. Following a high storm flow, benthic algae regrow and planktonic algae slowly accumulate downstream (Figure 4.2). Benthic invertebrates that graze algae can constrain algal accumulation if growth rates are low, but algae es-

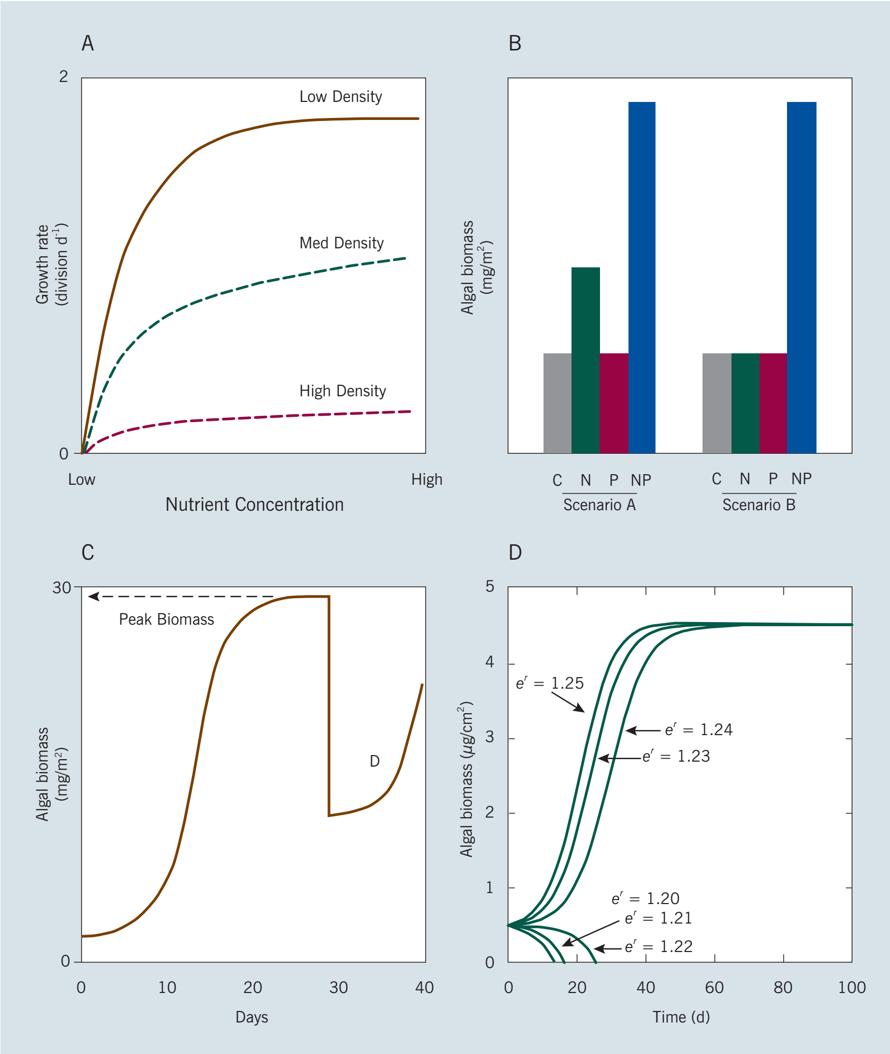


Figure 4.2:
Basic relationships between algal growth rates and nutrient concentrations. A) The asymptotic relationship between algal growth and nutrient concentrations, which decreases with algal density on substrata. B) Scenario A shows primary limitation of algal growth by nitrogen and secondary limitation by phosphorus. Scenario B shows colimitation by nitrogen and phosphorus. C) Benthic algal biomass starts out low after a storm event and grows to reach peak biomass in a 2-4 week period, after which it can slough from the substratum and then regrow. D) Results of simulation model showing sensitivity of algal accrual during assemblage development to slight changes in algal growth rate (e^r = 1.20-1.25) when herbivory is held constant

Source: Adapted from Stevenson (1997).

cape constraint when nutrients are high enough to produce growth rates that exceed grazing rates by invertebrates. The interaction of disturbance, algal recolonization, and potential constraint on algal accumulation by nutrient concentrations generates a threshold response in algal accumulation at the nutrient concentration that algae can outgrow grazing rates (Figure 4.2). As colonization time increases after a storm disturbance, the difference increases between algal accumulation in habitats with nutrients above and below the nutrient concentration threshold.

Effects of nutrient pollution on rivers vary with seasonal changes in light and temperature. Most problems with benthic algae in rivers are associated with filamentous macroalgae, such as the green alga *Cladophora* growing on rocks or the cyanobacterium *Lyngbya* growing in springs. The green alga *Cladophora* blooms when water temperatures are between 16 and 24°C (Figure 4.3). During cold seasons, diatoms are most abundant; and except for nuisance growths of some invasive species (such as *Didymosphaenia geminata*), diatoms are seldom a nuisance in streams. Planktonic algal blooms in rivers usually occur when low stable flows occur and nutrients are sufficiently high for algae to grow fast and accumulate. Most planktonic algal blooms are associated with warmer periods of the year, when rainfall is less frequent. Warm temperatures and nutrients stimulate algal growth, and warm temperatures favor the cyanobacteria. Many types of algae can cause taste and odor problems, as well as clog filters in water treatment plants, but the cyanobacteria can produce toxins that threaten human health.

Both nitrogen and phosphorus availability can limit algal growth in streams and rivers (Franceour et al. 2001). According to Liebig's Law of the Minimum, only one resource can limit growth and reproduction of a species at a time. With nitrogen and phosphorus being the most common limiting nutrient resources in rivers, three scenarios are possible for nutrient limitation (Figure 4.2B). In scenario A, algal growth is primarily limited by either low nitrogen or phosphorus concentration, and the other nutrient causes secondary limitation. In scenario B, increases in either nitrogen or phosphorus concentrations alone would not increase algal growth rates; concentrations of both nutrients must be increased to increase algal growth. In scenario C (not illustrated), both nitrogen and phosphorus concentrations are so high that increases in their availability would not stimulate further growth, so neither nitrogen or phosphorus availability limits algal growth. If nutrient concentrations are sufficiently low, either nitrogen or phosphorus would be limiting depending upon ratios of nutrient concentrations in the habitat. In most regions, phosphorus tends to be the most limiting nutrient. However, in regions with volcanic rock, nitrogen can be the primary limiting nutrient. In addition, terrestrial vegetation during the growing season can sequester sufficiently large quantities of nitrogen such that nitrogen may become limiting in streams.

Nutrient concentrations that limit algal growth vary greatly among species. Evidence from experimental streams and surveys of algal biomass in streams indicate a rule of thumb that peak algal biomasses are possible when total phosphorus is greater than 30 μg/L and total nitrogen is greater than 300 μg/L. In general, diatoms and most cyanobacteria have lower nutrient requirements

Figure 4.3:
A) A relatively natural occurrence of diatoms, the golden brown color on the stream bottom on either side of the storm-scoured central path in the middle of the stream and B) A nuisance growth of the green filamentous alga, Cladophora, *filling the stream*

than nuisance species of filamentous green algae. As algae accumulate in rivers, their densities can become sufficiently high that they deplete nutrient supplies. Thus nutrient concentrations in rivers that are higher than the 30 and 300 µg/L phosphorus and nitrogen concentrations can continue to cause greater algal biomasses because algae have sufficient nutrient supply to grow longer.

The relationship between biodiversity and nutrient pollution is more complex than single species growth-nutrient relationships. Nutrients negatively affect individual species indirectly by shifts in competitive hierarchies, grazer selection, and potential stimulation of bacteria, fungi, and viruses that cause disease in algae. Elevated nutrient concentrations make the habitat available for species requiring higher nutrients. Thus, the relationship between nutrients and algal biodiversity is a hump-shaped curve with a peak at intermediate nutrient concentrations. Low nutrient concentrations constrain which species can survive in the habitat and in high nutrient concentrations, habitats may be so altered physically and chemically by algal growth that some species of algae are not able to survive.

One of the critical questions in evaluating nutrient effects on algal biodiversity is whether species adapted to low nutrient concentrations are lost when nutrients increase from low to intermediate levels. In other words, as numbers of all algal taxa increase with increasing nutrients to intermediate concentrations because high nutrient taxa can invade, do we lose some highly sensitive taxa characteristic of natural, low nutrient conditions – our sensitive native species? Evidence suggests extirpation of diatom species in some streams as nutrient concentrations increase from low to intermediate levels. In large scale surveys of algae, we do not observe some taxa in intermediate and high nutrient habitats, even though these habitats were historically low nutrient habitats in which these taxa were characteristically abundant.

4.3. Nutrient effects on invertebrate and fish biodiversity

Invertebrate and some fish communities are strongly food limited in streams. Thus, nutrient driven increases in algal production have been observed to stimulate invertebrate and fish abundances. Fish and invertebrate biomass has been observed to increase two- to more than ten-fold in nutrient enriched rivers, and at large spatial scales, their biomass in rivers has been linked to phosphorus concentrations (Peterson et al. 1993). However, increased biomass does not mean increased biodiversity. In fact, negative effects of nutrients on invertebrate and fish biodiversity have been observed, especially in headwaters and wadeable streams. Nutrient enrichment leads to a decrease in pollution sensitive fish species, insectivores, and top carnivores, while omnivores and tolerant species increase. Similarly, carnivorous invertebrates and other pollution sensitive taxa decrease with nutrient enrichment as omnivorous invertebrates and tolerant species increase. State-wide surveys of fish and invertebrate biodiversity in the United States indicate that many attributes of biodiversity are negatively affected by nutrient concentrations. Independent results in West Virginia, Ohio, and

Algal excess and oxygen: An apparent contradiction

Box 4.2

Low dissolved oxygen in streams is caused by nutrients when they stimulate growth of autotrophs. This is often a perplexing relationship to understand, because we think most about how algae or aquatic plants add oxygen to streams. This is true, but algae and plants, like all other organisms, also respire to get energy that fuels metabolic reactions in cells. Those metabolic reactions make the proteins, lipids, and carbohydrates needed for cell function. So during respiration, oxygen is used and carbon dioxide is produced as a waste. The waste products of cell respiration, carbon dioxide and water, are used with light by algae and plants in photosynthesis, to produce sugars and oxygen. This cycling of carbon, hydrogen, and oxygen back and forth in the forms of carbon dioxide and water versus sugars and oxygen is one of the great balances in nature that occurs within a river and within our entire biosphere. The oxygen produced by autotrophs during the day can increase the oxygen concentration in streams. During both day and night they respire and use dissolved oxygen. Therefore, oxygen concentration in streams increases during the day if there is more photosynthesis than respiration, but it decreases at night because photosynthesis does not occur in the dark, just respiration. After a storm disturbance, algae and plants start regrowing, and as they accumulate day-night fluctuations in dissolved oxygen increase. When we add nutrients to rivers, autotrophs grow faster between storm events and thus fluctuations in dissolved oxygen

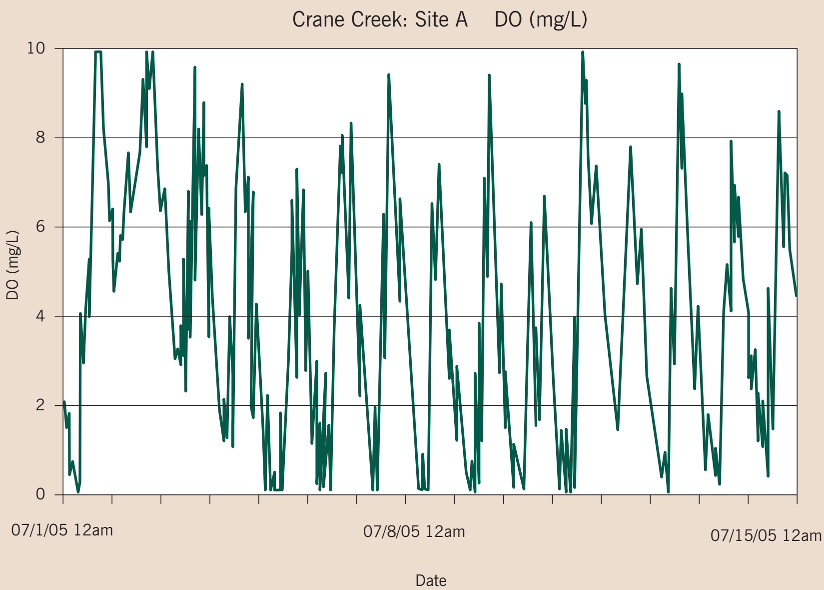


Figure 4.4: *Fluctuations in dissolved oxygen (DO, measured in milligrams/liter) during 24 hour cycles of light and dark periods in Crane Creek, a tributary of Lake Erie in the USA (figure courtesy of Michael J. Wiley, The University of Michigan). A storm disrupted algae on 7/1/2005. Afterwards algae and other biota in the stream regrew and produced greater and greater diurnal fluctuations in dissolved oxygen. Note how dissolved oxygen decreased to zero for longer periods of time on 7/7 and 7/12 in the early morning hours after previous light periods in which daytime oxygen concentrations stayed relatively low, perhaps caused by cloudy days*

Box 4.2 *(cont.)*: Algal excess and oxygen: An apparent contradiction

have greater amplitude and extend over longer periods of time. When fluctuations are really great, all oxygen in the stream can be used at night. Occurrence of these low oxygen events is difficult to predict, because they happen under relatively unusual weather patterns. But, when low oxygen conditions do occur, they can kill many organisms in the stream. This is one cause of fish kills in rivers. High pH, like low dissolved oxygen concentration, also stresses aquatic organisms and results from excess algal accumulation in a habitat. When algae photosynthesize, they consume carbon dioxide, which increases pH because of the role of carbon dioxide in a chemical equilibrium with carbonic acid and carbonates.

Wisconsin indicate that nutrients should be limited to less than 60 µg TP/L to protect the biodiversity of fish and invertebrates in their streams (e.g. Miltner and Rankin 1998).

Dissolved oxygen stress is the most commonly cited cause of loss in fish and invertebrate biodiversity with nutrient pollution, but physical habitat alterations by high algal accumulation and elevated pH are also issues. Excess growths of algae and associated bacteria can reduce oxygen concentrations and increase pH in streams. Dissolved oxygen is a limiting resource for fish and aquatic invertebrates. Government agencies around the world establish dissolved oxygen criteria between 4 and 6 mg/L and pH criteria of 9-10 to protect fish and invertebrate biodiversity. Lower dissolved oxygen concentrations than these criteria can be lethal to many species of fish and invertebrates (Davis 1975). Fish and invertebrate behaviour and reproduction are even more sensitive to lower dissolved oxygen than their death.

Response of fish and invertebrates to reduced dissolved oxygen varies greatly among species. The oxygen affinity of blood varies greatly among species. Many species of invertebrates don't have hemoglobin in their blood, so they have very limited affinity for oxygen circulation through their bodies and are more sensitive to low oxygen. Invertebrates have a great diversity of respiratory adaptations, ranging from gills, cutaneous respiration, and anal siphons. Anal siphons (tubes) allow mosquitoes to obtain oxygen from the air, which is why they bob with their bottoms toward the surface of the water. Top carnivores are probably highly sensitive to reduced dissolved oxygen because their bodies tend to be bigger and they have to be active to get their food. Variability in sensitivity among fish and invertebrate species to physical habitat alterations and pH has also been noted, but they have not been studied as thoroughly as oxygen sensitivity.

4.4. Challenges with measuring biodiversity responses to nutrients

Nutrient pollution exacerbates the challenging problem of estimating algal and invertebrate biodiversity in rivers. When hundreds of species occur in a habitat, tens of thousands of organisms must be observed to estimate the number of species in a habitat. Nutrient pollution increases the growth rates of species that require high nutrient concentrations. These high nutrient species often have very high maximum growth rates, resulting in very uneven abundances of species, with the rapidly growing high nutrient species having highest abundances. Uneven abundances of species create challenges for measuring biodiversity in rivers because more species will be relatively rare and not observed using routine methods for sampling and sample analysis.

Management of biodiversity requires a clear definition of goals and how and why what we learn is related to those goals. One rationale for protecting biodiversity (case 1), which is consistent with endangered species protection, is to protect the regional loss of species, or in case of highly valued game fish (e.g. salmon), loss of viable populations of evolutionarily and genetically distinct breeding populations. Protecting biodiversity, defined in this way, protects a final ecosystem service in which we have moral and aesthetic reasons for protecting species. Another reason for protecting biodiversity (case 2) is to protect the function of ecosystem services in the face of environmental change (Cardinale and Palmer 2002). In this case, we only need enough taxa to protect ecosystem function and related provisioning services. And finally, we have the concept of biological integrity, as defined by Karr and Dudley (1981), "the capability of supporting and maintaining a balanced, integrated, adaptive, community of organisms having species composition, diversity, and functional organization comparable to that of natural habitats of the region." In the latter case (case 3), we manage nutrient pollution for minimally disturbed conditions and the species that characteristically occur in minimally disturbed conditions. Each of these definitions of biodiversity carry scientific challenges for measurement and quantitatively relating to nutrient pollution. In case 1 we need to measure all species in a habitat (true diversity), the species that are critical for supporting ecosystem function in case 2 (functional diversity), and a representative subset of all species that provide assurance that ecological conditions are minimally disturbed in case 3.

Exploring these scientific issues for protecting endangered species in rivers allows producing simplified models for exposing concepts in managing biodiversity. Let's assume that our goal is to protect algal species from regional extirpation. Our definition of extirpation of a microalgal and bacterial species from habitats is poorly understood. Let's say we are interested in whether an algal species is extirpated from a stream. What is extirpation? Gone? Zero? As a wise microbial ecologist once said, "It only takes one" (Francis Drouet, Philadelphia Academy of Natural Sciences 1975). Because algae and microbes reproduce asexually and any single cell can then transform into a specialized cell for sexual reproduction, the successful reproduction and growth of just one cell is sufficient to restore the population of a microbial species in a stream. Given that cell densities of benthic algae and bacteria are commonly one billion cells per square meter of stream bottom, the number of cells in a stream is very large. Our routine method for ecological characterization of algae (and invertebrates) is examination of 300 cells in a sample from a stream. More thorough examinations sometimes call for 10,000 individuals in a sample, but this method is not used often. We never examine all the organisms from a habitat (except maybe trees, but then we do not sample all the seeds). The fact is, that we could be losing many more species than we observe missing because we did not know they were there to begin with. The problem with thinking about conservation of the biodiversity of microbes is that we have a very poor assessment of the true diversity of species in a habitat.

Although our understanding of the nutrient effects on rivers is not perfect, river science is sufficient to set nutrient management targets

In case 2 we are trying to estimate functional diversity, the identity and number of taxa that could grow and replace the function of lost taxa if environmental conditions changed. Functional diversity is also difficult to assess, but at least more practical than the true diversity of case 1. Modeling helps us understand requirements for assessing functional diversity. If we assume that we are trying to identify the species that could accumulate over a specific time period to replace ecosystem function of the dominant taxa, we need four pieces of information for the model: the length of time that species should have to replace the function of dominant taxa; the potential growth rates of the replacement taxa; abundance of dominant taxa (e.g. cells/cm^2); and the abundance of all cells (e.g. cells/cm^2) in the habitat of interest. Then, using the simple growth equation

$$N_t = N_0 e^{rt}$$

(where N_t and N_0 are the number of cells per unit area at time t in the future and time 0, the beginning; r is the growth rate (per day); and t is a number of days in the future)

we can estimate the number of cells that we would have to identify in a sample from the habitat to estimate functional diversity. We will assume: growth rates of algal cells in rivers are commonly 0.25 divisions d^{-1}; abundances of benthic algae are between 1-10 million cells per cm^2 of substratum; and we will allow one month for rare species to recover and replace abundances of dominant species. Given these assumptions, one cell could accumulate to be about 2,000 cells in a month. It would take about 1,000 cells to accumulate to 2 million cells in 30 days and replace the function of an extirpated dominant species. If we had between 1 and 10 million cells in the habitat per cm^2, then we would have to examine between 1,000 and 10,000 cells to detect any species with 1,000 cells/cm^2 on day 0, which according to our model are species that could replace function of dominant taxa over a 30 day recovery period. If however, we allowed 60 days for recovery, which is a typical period of relatively consistent ecological conditions (a season) for algae in a river, then one cell could accumulate to be over 3 million cells with the same 0.25 division per day. To identify all algal taxa that could accumulate over a 60 day period and replace the function of past dominant taxa, given conditions as described, we would have to examine 3,000,000 cells. Thus, it is practical to estimate functional diversity of algae in rivers, but it will require more extended analyses of species composition of algae than we currently employ.

Practical management solutions can be applied to reduce and treat nutrient pollution to protect instream and downstream biodiversity and uses of water bodies

We have similar problems for characterizing biodiversity of aquatic invertebrates, plus an additional problem. First, the diversity of invertebrates in a habitat is very high; so observing most of the species in a habitat would require a large effort. In addition, we seldom evaluate species level occurrences of aquatic invertebrates in surveys, which is needed to inform assessments of endangered species. Many invertebrates are immature insect stages in rivers and many of those cannot be identified to species level. Most monitoring of aquatic invertebrates involves identifications of genus and higher levels of taxonomy. For algae and aquatic invertebrates, new molecular techniques offer the potential for high taxonomic resolution and high detection sensitivity. Fish and mussels are the two groups of organisms in aquatic habitats for which we can, with a level of accuracy appropriate for endangered species management, determine the presence and absence of species in a stream. Often the diversity of fish and mussel species is 30 or less in a habitat.

A practical solution for protection of biodiversity in rivers from effects of nutrients, given the challenges with measurement of true or functional diversity, is to manage rivers for ecological integrity, which is characterized by the physical, chemical, and biological condition of rivers that have very low levels of human alteration. This approach is based on a major tenet of conservation biology, that is, that preserving physical and chemical integrity of ecosystems will provide conditions for protecting biodiversity in that ecosystem. The methods for assessing physical, chemical, and biological conditions of rivers have been established and practiced in many parts of the world for ecological assessments that satisfy government regulations. These methods are becoming sufficiently accurate that the minimally disturbed condition for individual river segments can be predicted. They are also highly sensitive, such that modest changes in human disturbance can be detected. Thus, we can assess whether the biological integrity of Karr and Dudley (1981) is being met in a river segment and detect deviations from these conditions. If we assume that true and functional diversity are also protected, we have a reasonable and practical method for determining whether biodiversity of a site is being protected. Of course, it is possible that historic disturbances have caused extirpated species, but many lines of evidence suggest that rivers have great capacity for recovery if species are not regionally extirpated. If we protect the minimally disturbed habitats in which we observe many of the sensitive species that disappear with nutrient pollution and other stresses on river ecosystems, we are likely to protect the sensitive species that we have not observed.

4.5. Nutrient effects on ecosystem goods and services

Biodiversity is one of many goods and services provided by rivers and streams. Ecosystem goods and services are benefits to humans resulting from materials provided by or processes performed by ecosystems (MEA 2005). Obviously, rivers provide direct value to humans as a source of drinking water, which was historically relatively uncontaminated by human activities. Today, of course, most waters are contaminated by waste from humans that live upstream, so are unsafe for consumption without treatment or at least boiling. Even though great progress has been made toward goals of increasing availability of safe drinking water, over 600 million people are expected to lack that access in 2015 (UNEP 2012). In a very real way, transport of human waste away from their sources is an important service of rivers. Rivers, as well as associated wetlands and in-line lakes, are important for breakdown and transformation of those wastes into less toxic forms as well as their entrainment. Although waste transformation and transport are not ecosystem services for which economic markets exist, these services have value indirectly through other goods and services that result, such as cleaner downstream drinking water and sustainable fisheries.

Ecosystem goods and services have been grouped into four categories: provisioning, cultural, regulating, and supporting services. Water for drinking, irrigation, and industry, fish and shellfish, and hydropower are examples of provisioning services that have direct effects on human well-being and for which markets are commonly established. Cultural services are a bit more difficult to market, but they do have direct benefits for people. The aesthetic and recreational values of water for swimming, water sports, and fishing are examples of cultural services that have great economic importance. The support of biodiversity is also a cultural service from the perspective that for moral and spiritual reasons, people feel that protecting species is the right thing to do. Protecting species in ecosystems has also been related to protecting the sustainable functioning of ecosystems and the services they provide.

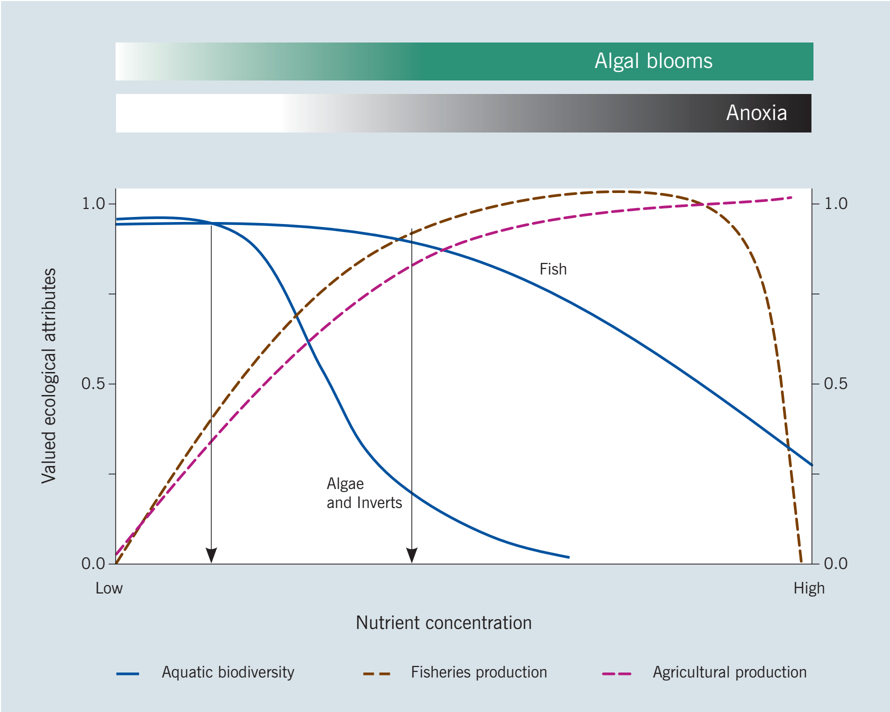
Provisioning and cultural services are referred to as final services, because they have direct benefits to humans. Two other categories of services, regulating and supporting services, are referred to as intermediate services because they do not directly benefit humans, but rather influence other services. Waste transport, biogeochemical transformation of wastes, organic matter processing, and nutrient cycling and retention are examples of regulating services. Regulating services transform ecological materials to mitigate leakage and disposal of wastes. Primary production, wildlife habitat, and resulting biodiversity can be considered supporting services, because they provide the resources for either regulating, provisioning or cultural services.

Effects of nutrient pollution on ecosystems services vary greatly among rivers in different geological, climatic, and economic settings. In addition effects of nutrient pollution present tradeoffs for managing rivers for different ecosystem services. Nutrient pollution negatively affects drinking water quality and most cultural services, likely including protection of sensitive taxa adapted to low nutrient concentrations (Figure 4.5). Most regulating services are positively affected by nutrient pollution, because increases in algal growth or nutrient concentrations would increase primary production, organic matter processing, and nutrient cycling. In addition, many provisioning services, for example fisheries, are positively affected by low and intermediate levels of nutrient pollution, but negatively affected by high levels.

4.6. Treatment and policy solutions for nutrient pollution management

Aquatic resource managers are faced with great challenges in nutrient pollution management because of tradeoffs in optimizing uses of ecological resources across regions. Tradeoffs are the fundamental challenge of managers (Ayensu et al. 1999). Tradeoffs occur at the scale of the habitat itself, with some ecosystem services of rivers being optimized at low levels of nutrient pollution and others at intermediate levels of nutrient pollution (Figure 4.5). Tradeoffs are compounded when resource uses of lands in a catchment are considered in the management plans that would optimize the uses of both terrestrial and aquatic ecological resources in a region.

Figure 4.5: *Tradeoffs among uses of rivers indicated by hypothetical relationships between a resource stressor (e.g. nutrient concentrations) and a suite of ecosystem services of catchments: drinking water quality; algal, invertebrate, and fish biodiversity; fisheries production; and agricultural production. The vertical lines indicate nutrient criteria that could be used to protect different uses in different waters*



Source: Modified from Stevenson and Sabater (2011).

Nutrient pollution is generated by many different alterations of a watershed by human activities. Relatively small amounts of nutrient pollution result from activities as simple as creating roads in a landscape or clearing vegetation from lands. This pollution results from several processes. First, clearing trees from land removes vegetation that sequesters nutrients. Removal of vegetation allows nutrients to leak from the catchment. Often, waste vegetation from logging operations releases nutrients as they decompose. Clearing trees from land and building roads can increase runoff of water and eroding sediments into streams. Increasing runoff and rates of groundwater percolation can also cause hydrologic instability in stream channels, which leads to stream bank failure and additional erosion. Sediments washing into streams carry large quantities of phosphorus relative to nitrogen. Groundwater carrying nutrients leaking from catchment to stream channels carries more nitrogen compared to phosphorus.

Application of fertilizers to agricultural lands and lawns in urban environments are major sources of nutrient pollution to rivers. Fertilizer runoff from crop-lands is a major source of nutrient contamination, and far exceeds runoff from pastures. We see evidence for this in much greater correlations in relationships between either nutrients or algal biodiversity in streams and the croplands versus pasture land in catchments. In fact, our greatest threat to future nutrient pollution is the added demands on agriculture (Seitzinger et al. 2010). Streams in more affluent neighborhoods have higher nutrient concentrations in them than streams in poorer neighborhoods because of ability of households to purchase fertilizer.

Wastes from humans and livestock are also major sources of nutrients. Wastes from humans and livestock are discharged to streams from either municipal or agricultural wastewater treatment plants, if these facilities exist, or directly through sewers, storm drains, or channels without treatment. Often manure or treatment plant sludge wastes are applied to both pastures and croplands for fertilizers, and in some cases as means to dispose of wastes rather than just fertilization. Wastes from humans and animals also enter rivers via runoff and groundwater when wastes come from isolated households with septic tanks or straight pipes into waterways. Some industrial processes also generate nutrient wastes as byproducts of processing large quantities of organic material. Pulp and paper mills and food processing operations are two examples. Organic wastes that accompany nutrients in human, animal, and some industrial wastes are particularly problematic because they also contribute to low oxygen in rivers, potentially synergistically with nutrients.

Many options exist for reducing and treating nutrient wastes, with some providing options for sustainable biofuels. Vegetated riparian buffer strips provide substantial reduction in phosphorus runoff from croplands, with benefits observed in improved algal biodiversity (chapter 9). Agricultural fertilizer waste could be reduced by educating farmers about determining fertilizer needs and the small benefits of over fertilization, testing nutrients in soils, taxing fertilizers, and developing better risk-distribution so farmers do not over fertilize to ensure they get a good crop. Waste-water treatment plants are being developed with advanced nutrient removal technologies. Problems remain however, in costs of implementing these technologies relative to perceived benefits. In fact, most costs are probably overestimated and most benefits are underestimated. Costs may be overestimated if some expenses can be recovered by using wastes to produce beneficial products, such as biofuels. Organic wastes can be used to produce methane and ethanol in anaerobic digestors. Nutrient by-products from anaerobic digestors and treatment facilities can be used to grow algae, which can also be used in biofuels. Nutrient wastes become a valuable commodity when they are linked to energy production, which could lead to a long-term sustainable solution to nutrient pollution.

Benefits of ecosystems goods and services are generally not appreciated by the public. But that is often because they are not informed about protecting the services and the values of services to them. The value of protecting biodiversity for many people in the world is high. This can be quantified as a direct benefit for the moral and aesthetic value of biodiversity to the public. The value of protecting biodiversity could also be estimated for increasing efficiency and sustainability of final ecosystem services, if we could quantify those relationships better and when we relate improved efficiency and sustainability of final ecosystem services to their values.

So what could the value of ecosystem goods and services of rivers be, and how are they impacted by nutrient pollution? These numbers are difficult to quantify for a variety of reasons, but approximations have been made. Economic implications of nutrient pollution for human health have not been estimated, but a recent assessment of damages to recreation, property values, and drinking water conservatively estimated damages between 2.2 and 4.6 billion US dollars per year in the United States alone (Dodds et al. 2009). Economic losses to boating and angling (US\$0.37 to US\$1.16 Byr^{-1}) and lake property values (US\$0.3 to US\$2.8 Byr^{-1}) were estimated to be particularly severe, followed by costs associated with contaminated drinking water (US\$0.81 Byr^{-1}), and mitigation of biodiversity impacts (US\$0.04 Byr^{-1}) (Dodds et al. 2009). These estimates are criticized by resource economists because they double count values of final and intermediate services; i.e. if the direct value of an intermediate ecosystem service is through the value of the final ecosystem service it regulates or supports, then summing values of intermediate and final services would be double counting. In addition, the methods of valuation are questioned because they assume that values of ecosystem services do not differ across landscapes. However, these numbers are sufficiently high to illustrate the great value of rivers and damages caused by nutrient pollution. Ecosystem service valuation will actually be very important factors in management strategies as weights in social preferences for acceptable risk for losing one ecosystem service versus another.

4.7. Management targets for nutrient pollution

Given we know relationships between nutrient concentrations, biodiversity, and other ecosystem services, and we can have technologies that can reduce and treat nutrient wastes, how low do we need to reduce nutrient pollution and where? What should nutrient management targets be? How and why would these nutrient management targets vary among rivers? How can we achieve these management targets? Significant advances in river science and ecology as well as nutrient treatment and environmental policy allow us to answer these questions better today than 10 years ago – and implement those answers in environmental policy.

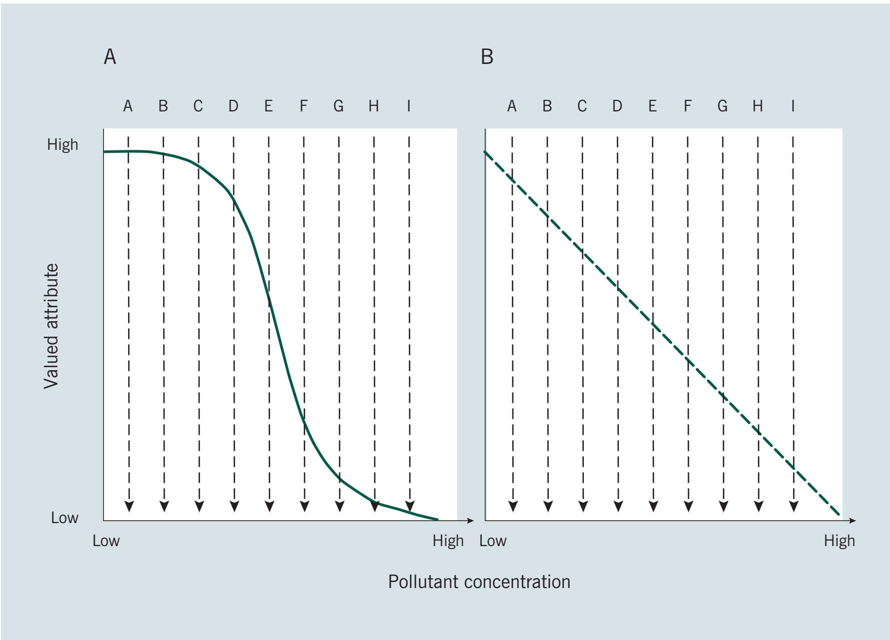
We can think of nutrient management targets as nutrient concentrations that provide an acceptable risk for sustaining an ecosystem service. In the US, these concentrations are referred to as nutrient criteria, which are part of water quality standards and related to protecting the specific uses of a waterbody. The designation of water body uses and related water quality criteria are codified in the rules of the Clean Water Act of the United States. Many other countries have similar laws and rules in which goals for pollution reduction are related to water resource uses, ranging from the European Union's Water Framework Directive to China's Water Pollution Prevention and Control Law. Historical application of the term "use" in its regulatory context is very similar to ecosystems services. Examples of regulatory uses of waters are drinking water, navigation, recreation, irrigation water, and aquatic life support (which is basically aquatic biodiversity). Thus, nutrient management targets are related to the uses of waterbodies.

Nutrient management targets are established in three different ways. One is to determine the lowest nutrient concentrations in a region where climatic and geological conditions are relatively similar and then apply those concentrations as targets for all waterbodies. This method is appropriate if large proportions of waterbodies are polluted in a region, if the pollution produces unacceptable changes in ecosystem condition, and if restoration of nutrient pollution to the lowest concentration in a region would provide benefits. Another method is to use characterizations of nutrient conditions at sites known to be meeting uses or meeting definitions of minimally disturbed (often called *reference*). Often the 75th percentile of nutrient concentrations at these sites provides appropriate upper bounds for a long-term average condition that will protect uses in similar waterbodies, but this approach does not specifically link nutrient concentrations to a problem. The last method is to explicitly and quantitatively relate nutrient concentrations to changes in measures of uses, determine desired level of uses, and use a model to determine the nutrient concentrations that provide the desired level of uses. The latter method is referred to as an effects-based approach (Figure 4.5). The effects-based approach is valuable because it explicitly relates use and contamination and it provides a means of evaluating tradeoffs among uses.

Thresholds in use-nutrient relationships are particularly valuable for establishing pollution criteria because they help develop stakeholder consensus (Muradian 2001). If a threshold relationship is observed in a valued attribute of an ecosystem, then the public tend to agree on a level of pollution that is acceptable for protecting a use. With threshold relationships the level of the valued attribute that is considered satisfactory is no longer a point of contention because the likelihood of protecting the valued attribute is either high or very low at different pollution levels, and presumable very low is unacceptable. Also the level of risk of losing the attribute is less a point of contention, because the range in pollution levels at which the valued attribute goes from high to low is very narrow.

The graph in Figure 4.6 provide an opportunity to explore the value of thresholds in relationships between a valued attribute and pollution concentration for environmental policy. Take this simple quiz. Assume that Figure 4.6A shows the relationship between something we really care about (life savings, happiness of our children, biodiversity) and a "pollutant" (volatility in stock markets, global strife and inequality, nutrients). What is the maximum level of pollutant to which you would be willing to expose your valued positions or feelings? Would you choose stressor level A, B, C, D, E, F, G, H, I? Most people pick B or C. Remember this is losing something that you really care about. Would you be willing to lose 5-10% of it? What if there was uncertainty about the level of stressor that occurs from year to year? That would likely cause you to pick even lower levels of pollution. In real ecosystems, there is uncertainty from year to year. If we

Figure 4.6: *A) Threshold responses in a valued ecological attribute, or bad attribute, help stakeholders develop consensus on appropriate pollution levels for protecting ecosystem. B) A linear response in a valued attribute along a stressor gradient*



only want to allow modest reductions in a valued attribute every 5-10 years, then B becomes the answer more than C. Curves with some assimilative capacity like Figure 4.6A allow for some stress in the system before collapsing. Figure 4.6B shows a relationship with a linear response, in which agreement is much more difficult because there is no one level of the pollutant that has a substantially lower effect than a slightly higher value of the pollutant. The challenge with a linear responses is that either no pollution is allowed, or the selected level of pollution allowance becomes difficult to justify to stakeholders with a diversity of opinions.

If tradeoffs in uses exist along gradients, then all uses of waters cannot be supported at optimal levels using the same nutrient management target. For example, if nutrient pollution reduces biodiversity but increases fisheries production (Figure 4.5), then is optimizing at low or high levels of pollution desirable? If we manage for intermediate levels of pollution, then we do not get optimal levels or potentially even satisfactory levels of either use. In fact, we may have lost considerable biodiversity at levels of nutrient pollution that provide fisheries and even moderate levels of agriculture or urban development in catchments. Different nutrient management targets must be used for different waterbodies to support all uses at satisfactory levels in one location or another. Low targets for nutrient management would protect biodiversity, water quality and recreational uses, but may not provide high productivity for fisheries or allow extensive agriculture in watersheds (Figure 4.5). Intermediate levels of nutrients have moderate risk to drinking water and recreational uses, but enable extensive agriculture in a watershed. If sufficiently low numbers of rivers are managed at intermediate levels of nutrient pollution, perhaps downstream uses could be protected as well. Allowing for different uses of different water bodies enables managing sets of rivers to protect all uses and achieving higher aggregate regional use benefits than by managing all waterbodies at the same level of pollution.

In addition to tradeoffs, another reason to manage waterbodies for different and site-specific levels of nutrient pollution is the impracticality of protecting all waterbodies for the low levels of pollution that would be necessary to protect sensitive species. First, the levels of nutrients that affect biodiversity in rivers are relatively low compared to concentrations observed in many regions of the world having even modest human alteration of catchments. Second, extensive contamination of soils and groundwater with nutrients makes restoration of some catchments difficult. Thus a reasonable strategy is to select one subset of all rivers to protect for uses related to biodiversity and drinking water and another subset of rivers could be established to protect uses for fisheries productivity and allow human alterations of landscapes at relatively extensive levels.

A minimum goal for all waters should be limiting nutrient pollution so that rivers continue to provide high levels of some ecosystem services and protect downstream conditions.

To achieve these goals for regional optimization, new questions emerge. Two questions are fundamental. What are the different uses for rivers in a region? How many and which rivers should be protected for the different uses? While it is beyond the scope of this chapter to address all factors associated with this question, we will show that the question can be addressed with sufficient accuracy that answers can be used for development of nutrient management policy.

First, a major issue for determining the number of rivers to conserve for different uses in a region is the values that regional people have for different ecosystem services. Valuation of ecosystem services varies internationally for a variety of factors, but particularly economic conditions. For example, greater value is placed on recreational and aesthetic conditions of rivers in affluent than poor regions. In many parts of the world, managing nutrients to protect biodiversity is not a priority for local or national governments. In fact, adding nutrients increases productivity for aquaculture which has great value for providing food in poor countries (Figure 4.7). Of course, the result is often more harmful algal blooms and low oxygen concentration in downstream rivers and lakes, which harm drinking water supply, human health, and fisheries. Integrated resource management can be used to evaluate the costs and benefits of different management strategies as well as identify who is responsible for damage and who should pay for restoration or lost resources, if that is necessary.

The question of how many rivers to manage for different uses also depends on the diversity of uses of rivers and surrounding ecosystems, tradeoffs among those uses, and acceptable risks for not supporting uses. For example, how many rivers must be managed to reduce nutrients to control hypoxia in coastal waters? Watershed models can provide a reasonable answer to that question for developing management strategies. How many rivers should be managed for recreational fisheries? Again, economic valuation of recreational fisheries can be estimated, as well as distance of rivers from potential users, which together could be used to develop an optimization model for river management. How many streams should be protected for species and which streams should be protected? Addressing these questions calls for understanding spatial meta-population dynamics and in particular, dispersal, colonization, and local extinction rates of organisms (Lowe 2002). Fewer habitats would need protection if dispersal rates, connectivity of habitats, and sub-population persistence are high. Because of the punctuated nature of low dissolved oxygen events in streams, maintaining high quality dispersal

Figure 4.7:
The stark realities of tradeoffs among uses of our waters are evident when traveling around the world. Nutrient management of waters in some parts of the world means adding nutrients to the water, rather than reducing nutrient pollution. Here are pictures of a farm in the Mekong River Delta of Vietnam. Manure from the pig is used to produce methane for cooking in a homemade anaerobic digestor. Then waste from the digestor is put into a canal to increase algal and bacterial production to grow fish as fast as possible. The fish adapt to the low oxygen concentrations in these waters by gulping air from the surface of the water

pathways for organisms may not be critical for those organisms if they can use contaminated pathways during low stress periods. At larger spatial scales, sets of streams should be selected from different climatic and geological settings as well as streams and rivers of different size because these streams would support the broadest diversity of organisms.

Scientists, policy makers, and other stakeholders are poised for major changes in environmental policy for nutrient management of rivers. Important instream and downstream problems are caused by nutrients in rivers, ranging from loss of biodiversity to impairment of drinking water, recreational, and fisheries uses. Whereas urban wastewater and agriculture have been major sources of nutrients in the last 50 years, increases in non-point source nutrients from agriculture needed to feed a growing world population likely present the greatest future threat to nutrient management. Solutions exist to minimize over fertilization and for nutrient harvesting in algal biofuels. Sound science will be critical and is available for developing nutrient management policy, as well as conceptual advances of linking science and policy. The great importance of these environmental problems to human well being calls for additional investment in science to refine solutions to these problems, but the lack of perfect knowledge is not justification for inaction. On the contrary, uncertainty in knowledge calls for greater caution and need for conservation. The time for action is now. We need local and national governments, as well as governments around the world, to cooperate on environmental policy. In particular, an internationally consistent nutrient management policy could protect biodiversity as well as other ecosystem goods and services in both instream and downstream waters. A policy, using site-specific goals for management, effects-based pollution criteria, and a long-term vision for achieving these goals could serve as a model for managing other cross-boundary environmental problems.

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Chapter 5

The River Drugstore: The Threats of Emerging Pollutants to River Conservation

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Due to the rapid growth of population, industrialization and change in life style, water contamination has become one of the major environmental concerns faced by the world today. This chapter focuses on pollution by organic micro-contaminants, giving an overview of the main sources and routes of entry of specific classes of contaminants, such as pesticides, pharmaceuticals, perfluorinated compounds, endocrine disrupting compounds, etc. Potential effects on aquatic organisms and humans are also discussed.

5.1. What is pollution?

Since early in history, people have dumped sewage into waterways, relying on natural purification by dilution and by biodegradation. However, the rapid growth of population, and change in life style resulted in a greater volume of domestic and industrial wastewater generated and subsequently discharged into the aquatic environment. Consequently water contamination became one of the major environmental concerns faced by the world today. Water quality has a direct impact on citizens and economic sectors that use and depend on water, such as agriculture, tourism, industry, energy and transport. It also affects river-associated ecosystems and the biodiversity they host. The effects of water contamination on humans are many, including disruption of the natural food chain, diseases, as well as serious harm to aquatic ecosystems.

There are many different types of water pollution and all have specific adverse effects on the environment and humans. The following types of water contaminants are usually distinguished:

— nutrients (nitrogen and phosphorus) and organic matter causing an increase in algal production and depletion of oxygen from the water column,
— inorganic compounds (salts, such as chlorides, sulphates and metals) that are toxic to aquatic organisms and can affect the rest of the food chain,
— organic (micro-) pollutants that can affect the health of aquatic organisms and those who eat them,
— microorganisms (virus, bacteria, protozoa) producing infectious diseases in aquatic life or terrestrial life through drinking water,
— suspended particles that reduce the amount of sunlight penetrating the water, and the oxygen transport into the sediments,
— physico-chemical pollution, such as thermal pollution (increase in water temperature due to hot water discharges) or change in the pH (i.e. acidification),
— radionuclids.

This chapter focuses on the pollution by organic micro-contaminants, a large group of compounds differing in their toxicity, mobility and behaviour in nature, and one that is constantly increasing in numbers, as new products are being synthesized and used in the industry. The chapter also gives an overview of the main sources and routes of entry of specific classes of contaminants, as well as potential effects on aquatic organisms and humans.

5.2. Sources

Generally, pollution of the aquatic environment originates from point or diffuse (or non-point) sources. *Point source pollution* enters a water body at a specific site, such as a sewer, and can generally be readily identified. Potential point sources of pollution include effluent discharges from wastewater treatment plants (WWTP) and industrial sites, power stations, landfill sites, fish farms, and oil spills from pipelines. Amongst them, WWTP deserve special attention as their effluents are the main source of many organic contaminants as shown in Box 5.1. Point source pollution generally can be prevented or at least reduced, since it is possible to identify where it is coming from. Therefore, the responsible person or agency can take immediate remedial action or invest in preventive measures such as longer-term investment in treatment and control facilities.

Diffuse pollution occurs where substances are widely used and dispersed over an area as a result of land use activities such as agriculture, farming and forestry.

Point source – Wastewater treatment plants (WWTP)

Box 5.1

WWTP effluents are the principal source and route of entry of many contaminants into the environment. Sewage is generated by residential, commercial and industrial establishments. It includes household waste liquid from toilets, baths, showers, kitchens, sinks and so forth that is disposed of via sewers. In many areas, sewage also includes liquid waste from industry and commerce. Sewage may include stormwater runoff in the case of combined sewer systems. These systems are usually avoided because precipitation causes widely varying flows reducing sewage treatment plant efficiency. With increasing urban population, changing lifestyles and industrialization, the quality of wastewater has deteriorated over the years, and hence requires treatment before it can be discharged into the aquatic environment or recycled for any purpose. In addition, the cost of water increases and environmental regulations for wastewater discharge become more stringent, thus making it necessary to implement more efficient treatment, including different physico-chemical and biological steps. Since the early 1970s until about 1980, aesthetic and environmental concerns were mainly considered, and wastewater treatment facilities were designed to reduce organic matter and nutrients such as nitrogen and phosphorus, and suspended solids. Since 1980, focus on health concerns related to toxics has driven the development of new treatment technology. However, in spite of advanced treatment options, not all organic contaminants are removed during treatment, resulting in degraded receiving water quality. Many toxic substances can

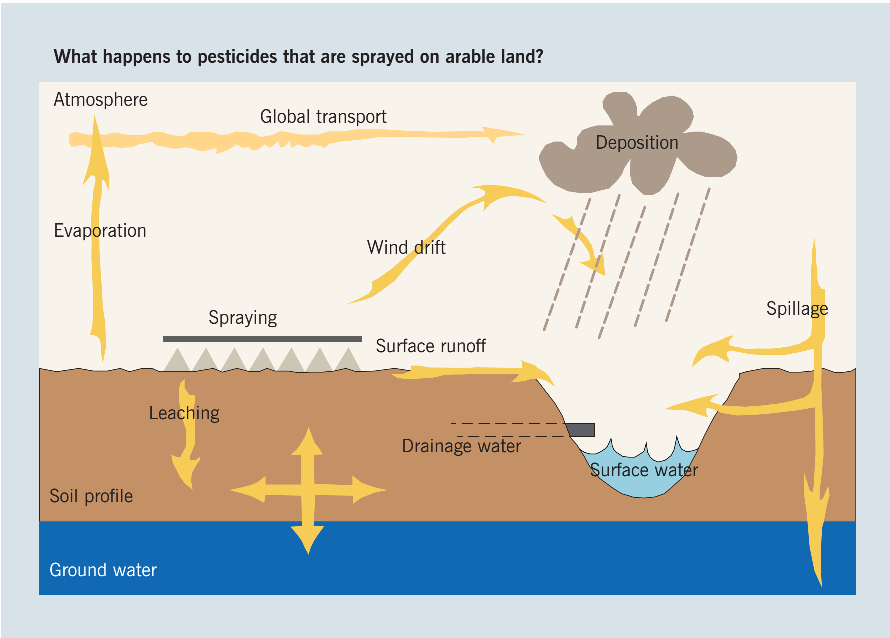
Figure 5.1:
Discharge pipe

Box 5.1 *(cont.)*: Point source – Wastewater treatment plants (WWTP)

pass through conventional treatment systems (based on activated sludge treatment). These are designed and dimensioned to achieve the prescribed removal of organic matter and nutrients, and their discharge is regulated on a national level to limit the total load to the recipient systems, thereby minimizing potential problems with oxygen consumption and eutrophication. However, no such regulations exist regarding organic micropollutants in the effluents from WWTPs. Most concerning are polar compounds that may occur in WWTP effluents because they are truly persistent under these conditions. The occurrence and removal of many polar chemicals has been studied, but still it is often not clear whether a certain class of compounds is widespread in municipal wastewaters, to which extent they are removed in WWTP and whether yet unknown polar metabolites are being formed.

Examples of diffuse pollution include the leaching to surface water and groundwater of contaminants from roads, manures, nutrients and pesticides used in agriculture (Figure 5.2) and forestry, and atmospheric deposition of contaminants arising from industry and combustion. It is often difficult to identify specific sources of such pollution, and prevention often requires major changes to land use and management practices.

Figure 5.2: *Diffuse source – Agriculture. Pesticides sprayed on fields can be transported to nearby surface water by run-off of rainfall or to groundwater by leaching of water through the soil. They can also be transported away in the air through wind drift and evaporation and then carried back to soil or water*



Source: Adapted from Swedish University of Agricultural Sciences.

5.3. Main classes of organic microcontaminants

Current use of chemicals by our technological society can be estimated in some hundreds of thousands of compounds (most of them organics) and this number is continuously growing (Figure 5.3). Depending on their properties and extent of use, a large amount of different chemicals can potentially reach the environment, their environmental and health effects being hard to predict in the long term.

The use of several products of unknown environmental impact was widespread after the Second World War, most used as products against agricultural pests, others applied as industrial products. Amongst them, DDT, aldrin, or parathion, but also heavy metals such as mercury, cadmium, or lead were used without any environmental protection. Public awareness of the effects of these products on the environment and human health was slow, it came by the hand of activists such as Rachel Carson and her renowned book *Silent Spring*. Non-polar hazardous compounds, i. e. persistent organic pollutants (POP) and heavy metals were in the focus of the administrations' interest and some of these products were

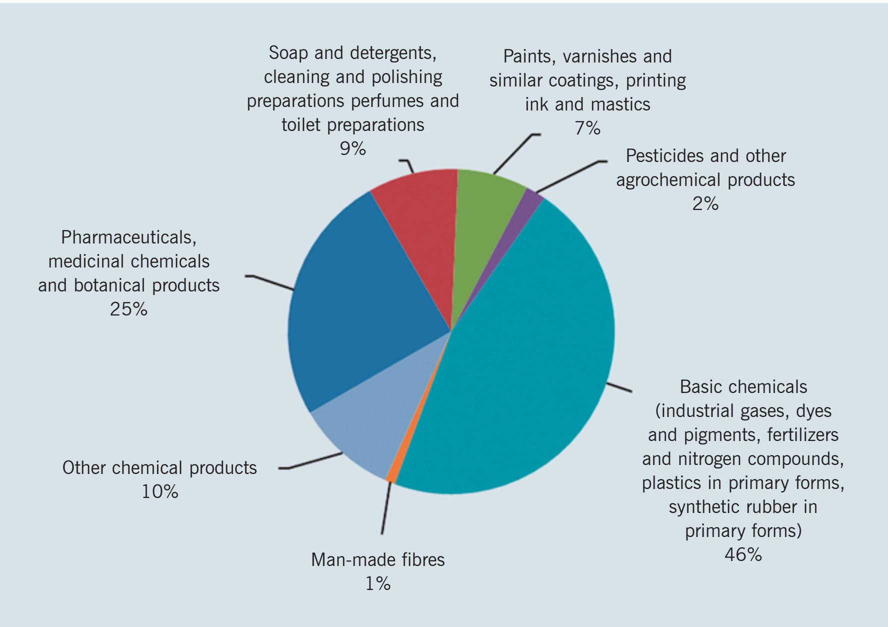


Figure 5.3: *Use of chemicals in our technological society. Manufacturing of chemicals and chemical products by production value in EU-27*

Chemical Abstracts Service (CAS): ~8,400,000 registered compounds.
European Union: ~100,000 compounds available.
REACH (EU Regulation on Registration, Evaluation, Authorisation and Restriction of Chemicals): ~30,000 compounds (10,000 already registered).
Source: Eurostat.

banned. The main concern is related to their persistance in the environment since they are resistant to environmental degradation through chemical, biological, and photolytic processes. Consequently, they can bioaccumulate in human and animal tissue and biomagnify in the food chain having significant impacts on human health and the environment. Their definition as priority pollutants came together with intensive monitoring and control programs. The global concern of these pollutants was finally defined by the United Nations treaty, signed in Stockholm, Sweden, in May 2001. Under the treaty, known as the Stockholm Convention, countries agreed to reduce or eliminate the production, use, and/or release of an initial twelve chemicals or chemical groups, the so called "the dirty dozen" (Table 5.1).

Today, the pressure of increasingly strict regulations, the adoption of appropriate measures and the elimination of the dominant pollution sources has resulted in a drastic reduction of emissions as well as a decrease in their arrival to the aquatic ecosystems. In the industrial sector, the reduction in discharges of POPs, particularly characteristic of the chemical, paper, textile and food processing sectors, was initiated in the 1970s with reductions at source combined with the implementation, made compulsory by legislation, of efficient wastewater treatment plants.

However, the so-called "emerging" or "new" unregulated contaminants have emerged as an environmental problem and there is a widespread consensus that this kind of contamination may require additional legislative intervention. A wide

Table 5.1: *The* dirty dozen

Aldrin – Pesticide widely used on corn and cotton until 1970. Closely related to dieldrin
Chlordane – Pesticide on agricultural crops, lawns, and gardens and a fumigant for termite control
DDT – Pesticide still used for malaria control in the tropics
Dieldrin – Pesticide widely used on corn and cotton until 1970. A breakdown product of aldrin
Endrin – Used as a pesticide to control insects, rodents, and birds
Heptachlor – Insecticide in household and agricultural uses until 1988
Hexachlorobenzene – Pesticide and fungicide used on seeds, also an industrial by product
Mirex – Insecticide and flame retardant
Toxaphene – Insecticide used primarily on cotton
Polychlorinated biphenyls (PCB) – Widely used in electrical equipment and other uses
Polychlorinated Dioxins and Polychlorinated Furans – Two classes of "unintentional" pollutants, by-products of incineration and industrial processes

range of man-made chemicals designed for use in industry, agriculture and as consumer goods, as well as chemicals unintentionally formed as by-products of industrial processes or combustion, are potentially of environmental concern. The term “emerging contaminants” does not necessarily correspond to “new substances”, i.e. newly introduced chemicals and their degradation products/ metabolites or by-products, but refers also to compounds with previously unrecognised adverse effects on the ecosystems, including naturally occurring compounds. Therefore, “emerging contaminants” can be defined as contaminants that are currently not included in routine monitoring programs and which may be candidates for future regulation, depending on research on their toxicity, potential health effects, public perception and on their occurrence in the environment. Their concern has been raised as a consequence of the progress achieved in analytical chemistry. Increased sensitivity in mass spectrometry, as a result of more efficient ionization techniques and better detectors, has enabled detection of virtually any contaminant at a very low level.

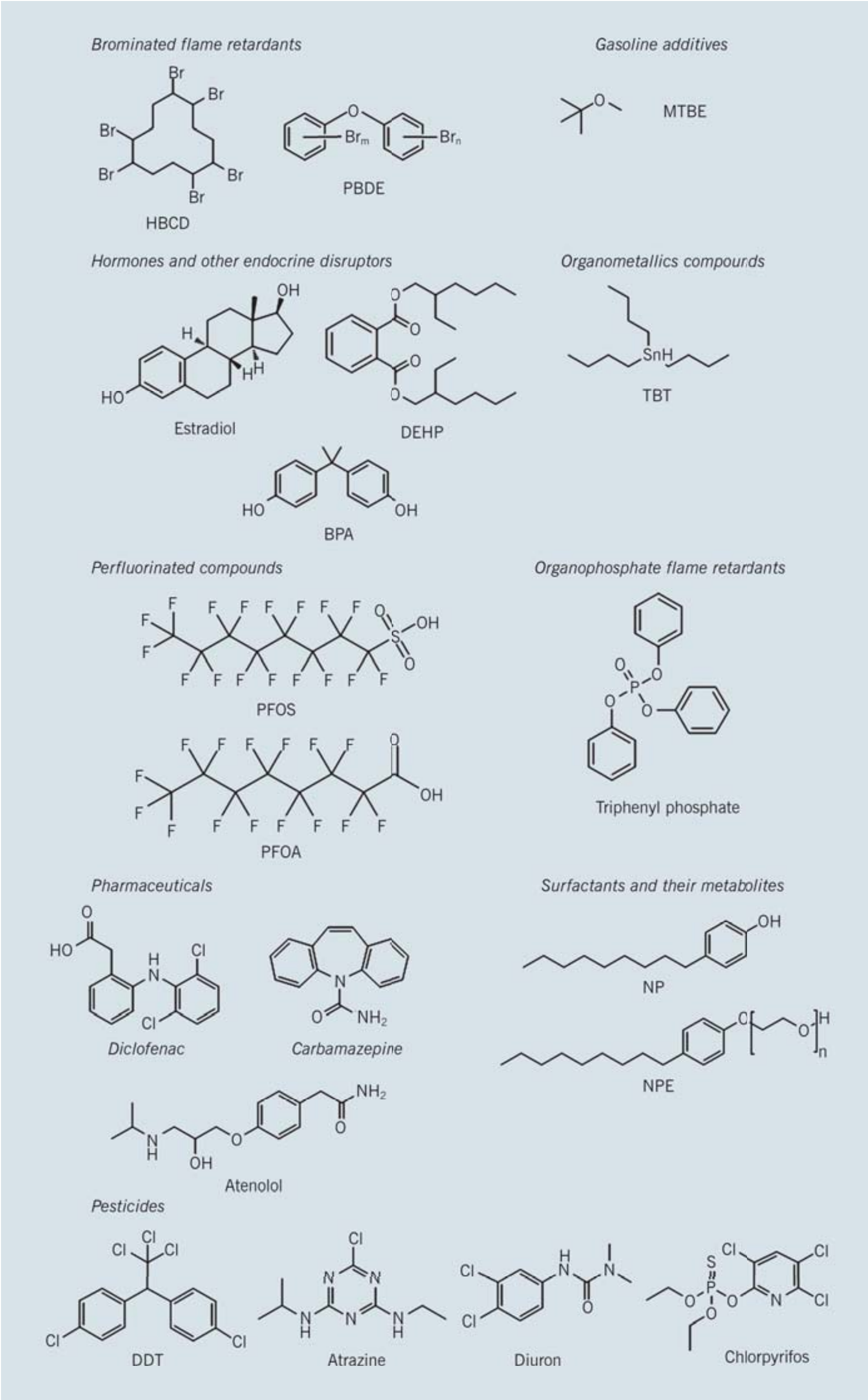
Particularly relevant amongst these emergent pollutants are several groups of compounds including brominated flame retardants, disinfection by-products, gasoline additives, hormones and other endocrine disrupting compounds, nanomaterials, organometallics, organophosphate flame retardants and plasticisers, perfluorinated compounds, pharmaceuticals and personal care products, polar pesticides and their degradation products, siloxanes, surfactants and their metabolites.

Some representative examples of common pollutants of the aquatic environment are shown in Figure 5.4. For most emerging contaminants, data on their occurrence and effects in the environment are not available and, therefore, it is difficult to predict what effects they may have on human health and on aquatic organisms. Numerous field studies are being conducted to identify the sources and points of entry of these contaminants into the environment, and to determine their concentrations in both wastewaters and the receiving environment.

5.3.1. Pesticides

Pesticide pollution in waters usually occurs from diffuse sources (runoff after application in agriculture), with minor point-type pollution from industrial emissions caused during production. Over the years the pesticides used have changed, from persistent compounds, hardly degradable, such as organochlorines (DDT, lindane, cyclodienes type aldrin, endrin, dieldrin, endosulfan, etc.) to more polar (water soluble) compounds that are degradable, such as N-methylcarbamate. However, their use has not stopped growing.

Figure 5.4: *Some representative examples of common pollutants of the aquatic environment*



Pesticides have been regulated and studied for decades. The current priority list of the EU Water Framework Directive (WFD) includes several individual compounds from different chemical classes, such as alachlor (aniline), atrazine and simazine (triazine), the chlorofenvinfos and chlorpyrifos (organophosphate), diuron and isoproturon (phenylurea), endosulfan (organochlorine), and trifluralin (dinitroaniline). All the above compounds are active ingredients of the phytosanitary formulations and their levels in the aquatic environment are certainly disturbing. However, the concern about pesticides is now focused mainly on their degradation products, which are often more ubiquitous and toxic than the parent compounds.

5.3.2. Pharmaceuticals and personal care products

The term "pharmaceutical" encompasses all prescription, non-prescription and over-the-counter therapeutic drugs, in addition to veterinary drugs and nutritional supplements. Personal care products include all consumer chemicals typically found in fragrances, lotions, shampoos, cosmetics, sunscreens, soaps, etc.

In the EU around 3,000 different pharmaceutically active compounds are used in human medicine. Most of modern drugs are small organic compounds, which are moderately water soluble, but still lipophilic, which permit them to be bioavailable and biologically active. They are designed to have specific pharmacologic and physiologic effects at low doses and thus are inherently potent, often with unintended outcomes in wildlife. Their consumption has increased recently and will continue to increase due to the expanding population, general ageing, increase of per capita consumption, expanding potential markets, new target age groups, etc. After they are administrated, pharmaceuticals are excreted via liver and/or kidneys as a mixture of parent compounds and metabolites that are usually more polar and hydrophilic than the original drug. Thus, after their usage for the intended purpose, a large fraction of these substances is discharged into the wastewater, unchanged or in the form of degradation products, that are often barely eliminable in conventional WWTPs. Depending on the efficiency of the treatment and chemical nature of a compound, pharmaceuticals can reach surface and ground waters.

Pharmaceuticals have been found in treated sewage effluents, in surface waters, in soil, and in tap water. Although the levels are generally low, there is rising concern about their potential long-term impacts to both humans and aquatic organisms as a result of the continuous environmental exposure to these compounds. These levels are unable to induce acute effects in humans, i.e. they are

Box 5.2

Case study: Occurrence of pharmaceuticals in the Ebro River basin

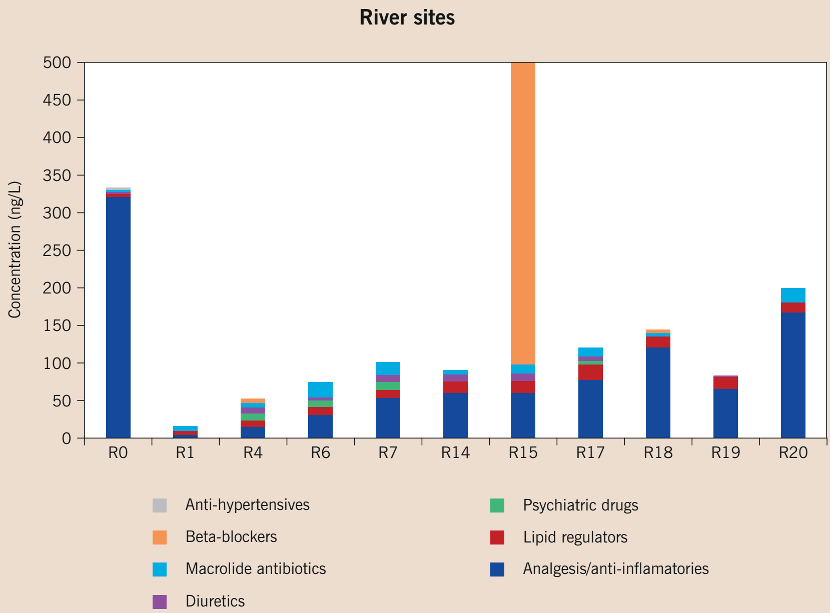
The Ebro River basin (northeast of Spain) drains an area of approximately 85,000 km^2, ending in the Mediterranean Sea and forming a delta of more than 30,000 ha. The most relevant economic activity in the region is agriculture (vineyards, cereals, fruit, corn, horticulture and rice production), but there are also some highly industrialized regions, mainly located in the northern-central part, close to the cities of Zaragoza, Vitoria, Pamplona, Logroño, Monzón and Lleida. Around 2,800,000 inhabitants live in the area.

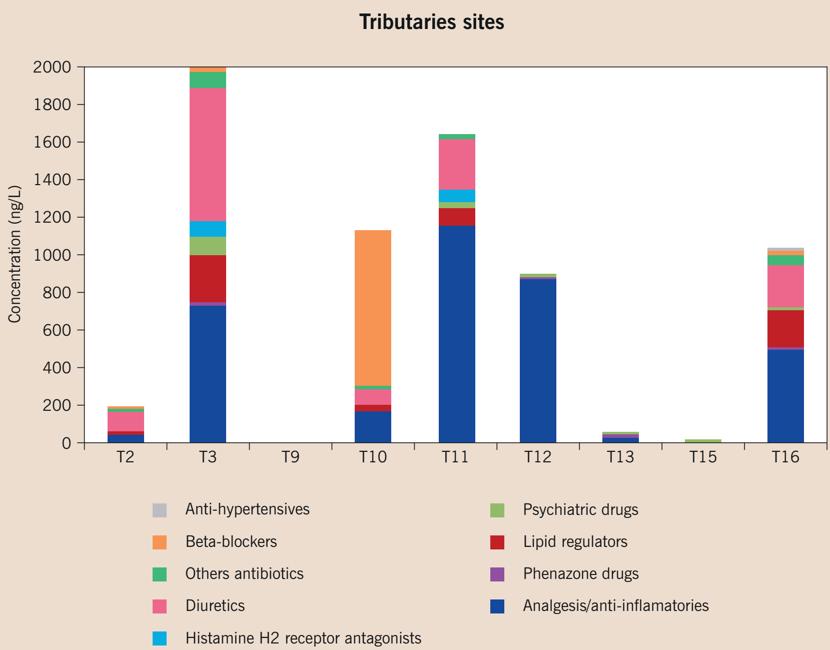
Pharmaceuticals were commonly found in the river water. The highest concentrations were detected in small tributary rivers adjacent to WWTP discharges (T3, T10, T11, T16), with low river flow rates and therefore low dilution of discharged effluents. The highest levels were found for anti-inflammatory drugs and analgesics such as acetaminophen, ibuprofen and ketoprofen. Other pharmaceutical classes showing high concentrations were the β-blocker atenolol and the diuretics hydrochlorothiazide and furosemide.

The hazard posed by pharmaceuticals in both surface and effluent wastewaters was assessed toward different aquatic organisms (algae, daphnids and fish). Studies showed that no significant risks exist in the Ebro River associated with the presence of pharmaceuticals, which indicate that reduction of compound concentration after wastewater treatment as well as dilution in the receiving river water efficiently mitigate possible environmental hazards. However, studies were only focused on the toxicity that individual compounds may cause to aquatic organisms, while toxicity of mixtures of pharmaceuticals (and also mixtures of other contaminants) was not taken into account. Therefore, in the absence of definitive data we should not relax our vigilance.

Figure 5.5:
The Ebro River basin, in N Spain, and the location of the sites in this case study

Figure 5.5 *(cont.)*





far below the recommended prescription dose, but have been found to affect aquatic ecosystems. Antibiotics and estrogens are among the many pharmaceuticals suspected of persisting in the environment either due to their resistance to natural biodegradation or to continuous release.

5.3.3. Natural and synthetic hormones

Female sex hormones and synthetic estrogens are considered the most potent endocrine disrupting compounds (compounds that interfere with the hormonal (endocrine) system in animals, including humans). Synthetic estrogens and progestogens are commonly administered in contraceptive formulations and for treatment of certain cancers and hormonal disorders as common as the menopause. Both natural and synthetic steroids, in either a conjugated (as glucuronides and sulfates, principally) or an unconjugated form, are excreted in the urine of mammalians and enter the aquatic environment via wastewater treatment plant effluents or untreated discharges. These potent estrogenic compounds have been shown to induce estrogenic responses in fish at concentrations in water (0.1-1 ng/L) (Purdom et al. 1994), concentrations often exceeded in the environment, and have been associated with certain alarming effects on reproduction and developmental processes, such as feminization, decreased fertility, or hermaphroditism (Colborn et al. 1993).

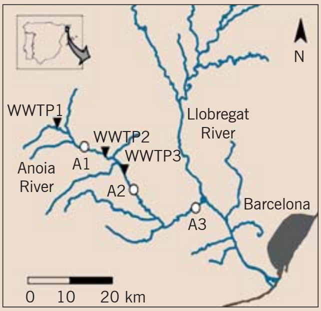
5.3.4. Alkylphenols

Alkylphenols, including nonylphenol (NP) and octylphenol (OP), are degradation products of surfactants alkylphenol ethoxylate type (APEOs). With a global production of about 500,000 tons per year, this type of surfactants are primarily used as detergents in both domestic and industrial sectors, especially in the textile, leather and paper industry, and as adjuvants for pesticides, paint ingredients, and wetting agents, among others. The concern lays in the fact that approximately 60% of APEOs entering WWTPs are released into the aquatic environment, 85% of them in the form of degradation products. Levels detected in the environment are in the range between ng/L and μg/L in water, sometimes approaching or exceeding those deemed to be sufficient to produce estrogenic effects, which are estimated in the case of the NP and NPEO1 between 1 and 20 μg/L. The most alarming and best studied effects are those that link exposure to these compounds in the aquatic environment with phenomena of feminization and intersex (the simultaneous presence of male and female reproductive organs) in fish (Solé et al. 2000; Petrovic et al. 2002, Box 5.4). This activity as endocrine disrupting compounds led to restrictions on their use in various countries, and the addition of NP and OP to the list of priority

Box 5.3

Case study: Occurrence of nonylphenolic compounds in the Anoia River (Llobregat River basin, NE Spain) and their effect on feminization and intersex of male carp

The Anoia River is a tributary of the Llobregat River and is situated in Catalonia, NE of Spain. The river receives effluents from several WWTP and is characterized by rather high levels of nonylphenolic compounds originating from WWTP treating effluents from several tannery and textile plants that use nonylphenolic compounds as surfactants. The study conducted in 2000 revealed a correlation between the presence of nonylphenolic compounds in water and sediment downstream of WWTP and plasma VTG concentration in male carp was observed. In female fish the synthesis of vitellogenin (VTG – an oviparous female egg yolk precursor) is regulated by estradiol (natural female hormone) levels in the plasma. In males, as a consequence of exposure to substances that mimic natural estradiol (in this case nonylphenolic compounds), VTG can also be synthesized inducing feminization in male fish and resulting in pathological intersex gonads formation (containing both male and female tissue).



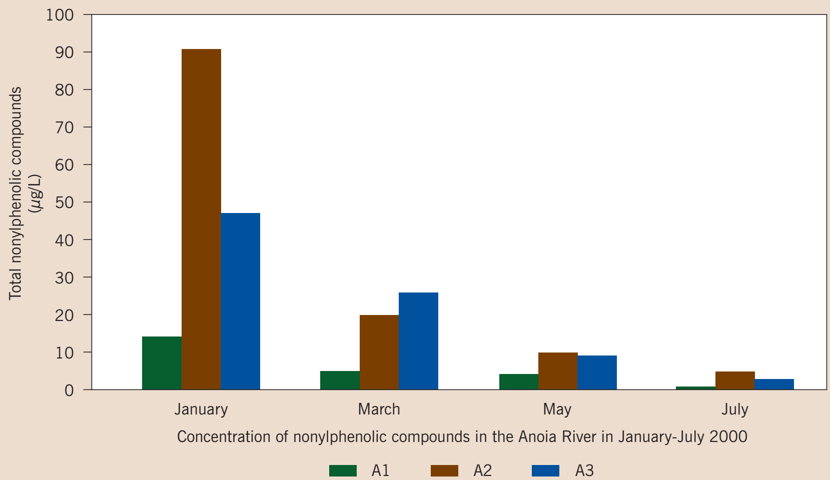
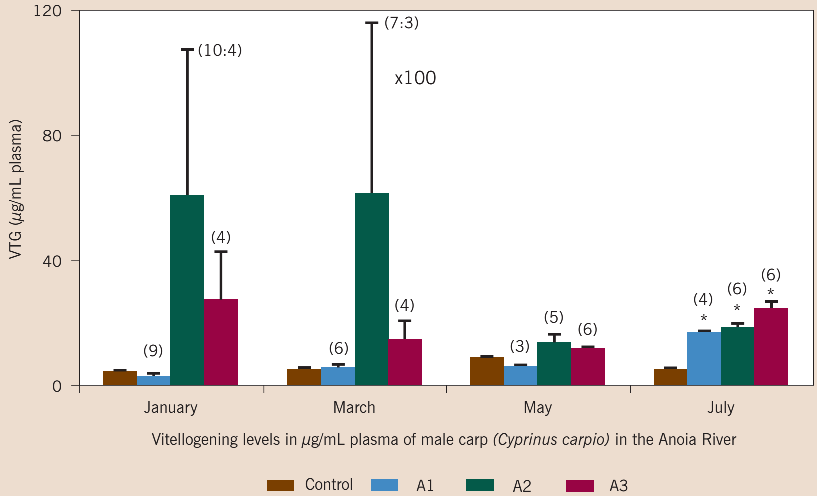


Figure 5.6: *Nonilphenolic compounds in the Anoia River and effects in carps*

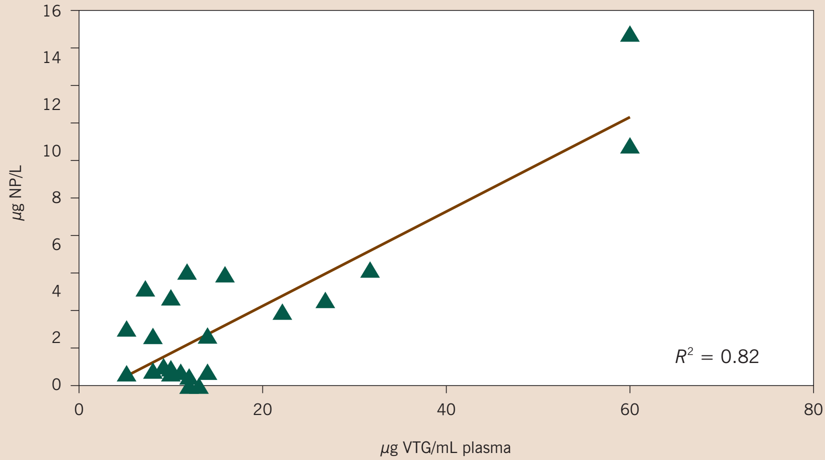
Box 5.3 *(cont.)*: Case study: Occurrence of nonylphenolic compounds in the Anoia River (Llobregat River basin, NE Spain) and their effect on feminization and intersex of male carp

Figure 5.6 *(cont.)*



Vitellogening levels in µg/mL plasma of male carp *(Cyprinus carpio)* in the Anoia River

Numbers in paretheses correspond to the number of fish analysed and the number of intersex fishes out the total



Correlation between VTG concentration in plasma an NP concentration in river water

substances in the EU. The average annual concentration of NP in surface water should not exceed 0.3 µg/L and the maximum should not exceed 2 µg/L. In the case of OP the maximum allowable concentration is not set, but the annual mean concentration should not exceed 0.1 µg/L in inland surface waters and 0.01 µg/L in other surface waters.

5.3.5. Perfluorinated compounds

Perfluorinated compounds comprise a large group of compounds characterized by a fully fluorinated hydrophobic linear carbon chain attached to one or more hydrophilic head. Perfluorinated compounds repel both water and oil, and are therefore ideal chemicals for surface treatments. These compounds have been used for many industrial applications including stain repellents (such as Teflon), textile, paints, waxes, polishes, electronics, adhesives, and food packaging (Clara et al. 2008). They are resistant to breakdown, and therefore persistent and bioaccumulative in the environment. They also biomagnify through the food chain.

There are many perfluorinated compounds, but some of the most studied are the following:

— PFOA or perfluorooctanoic acid, used to make fluoropolymers such as Teflon, among other applications.
— PFOS or perfluorooctanesulfonic acid, used in the semiconductor industry, and fire-fighting foam mixture.
— PFNA or perfluorononanoic acid, used as surfactant in the emulsion polymerization of fluoropolymers, like PFOA.
— PFBS or perfluorobutanesulfonic acid, used as a replacement for PFOS.
— POSF or perfluorooctanesulfonyl fluoride, used to make PFOS-based compounds.
— PFOSA or perfluorooctanesulfonamide.

The main direct routes of exposure of perfluorinated compounds to humans are in their diet and drinking water. They have been found in rivers, precipitation water, soils and sediments and biota samples. Perfluorinated compounds enter the environment through direct (directly from manufacture wastes or direct application) and indirect sources (due to their decomposition or disposal through product life cycle). WWTPs have been also identified as relevant pathway of their releases into the environment.

5.4. Effects on aquatic organisms and biodiversity

A fundamental characteristic of most biological systems is their diversity. The rapid loss of genetic, specific, and functional diversity resulting from anthropogenic disturbances (e.g. chemical stressors) is a significant environmental problem with global consequences (Chapter 6). Researchers and policy makers are becoming increasingly aware that species provide ecosystem goods and services that are essential for human welfare (Chapters 7 and 11). Contaminants exert their effects at all levels of biological organization, from molecules to ecosystems. Most research in environmental toxicology focuses on effects on lower levels of biological organization (molecules, cells, individuals) and is based in laboratory studies. This research improves our understanding of mechanisms of toxic action and exposure assessment, but must be linked with changes in communities and ecosystems. Following direct exposure to chemicals, sensitive individuals may die (i.e. lethal effect) or they may suffer sub-lethal consequences such as physiological, behavioural changes or impaired reproduction. Indirect effects may also occur, usually derived from direct effects on other species that induce changes in processes such as competition or changes in resource-consumer relationships. Direct toxic effects are easier to interpret than indirect effects, which received little attention.

Water contamination by man-made chemicals, chemicals unintentionally formed as by-products, nutrients and microorganisms, is one of the major environmental concerns faced by rivers today

One of the most important entry ways of contaminants into ecosystems is by ingesting organic materials. Materials and energy are processed and flow in ecosystems throughout food webs. Bioconcentration is defined as the uptake of contaminants directly from water, and bioaccumulation is the uptake of chemicals from either biotic (food) or abiotic (sediment) compartments. In aquatic organisms, bioconcentration describes the process which leads to higher concentration of a toxicant in the organism than in water. In the same way, bioaccumulation produces higher concentrations of a chemical in an organism than in its immediate environment, including food. Biomagnification refers specifically to the increase of contaminant concentration with trophic level. If biomagnification occurs, the concentration of contaminants increases with trophic level, i.e. concentrations in a consumer or predator exceed concentrations in the consumed prey. For example, periphyton and attached algae in streams concentrate contaminants several orders of magnitude above water levels. Organisms grazing these materials, such as several insect larvae (e.g. mayflies or midges), are exposed to significant concentrations that can be accumulated. Fish feeding on these contaminated insects may present elevated levels of a contaminant in tissues and consequently suffer a significant reduction in growth. The extent to which compounds accumulate and the routes that they are taken up and excreted may differ depending on the type of organism and chemical involved. These concepts are widely recognized as useful indicators of a biological exposure to toxicants.

Pesticides are designed to kill undesired organisms or pests, but these compounds also have direct and indirect (through trophic links) effects on non-target organisms. Herbicides target primary producers, such as algae, mosses or flowering plants, but they can also have effects on animals. Both algae and invertebrates suffer a loss in diversity and abundance, whereas species tolerant to pesticides can spread in rivers. For example, the herbicide diuron changes algae abundance and species composition in rivers, but at higher concentrations also produce lethal effects on invertebrates, tadpoles and fish. Additionally, sub-lethal effects can also occur at lower concentrations, close to those found in the environment. Diuron shows antiestrogenic and antiandrogenic activities in yeast, and these endocrine disrupting effects could change the fecundity of organisms and consequently have an ecologically significant impact on population dynamics. Similar studies with other herbicides also found changes in the community, from primary producers to herbivores and predators. Atrazine has negative effects on macrophytes, algae, and on the diversity and abundance of herbivore insects. In an experiment with the herbicide glyphosate at environmental concentrations, algal biomass increased as the herbicide killed herbivore tadpoles, and predator populations decreased as their prey died. Continuous exposure to pollutants can generate genetic adaptation in aquatic organisms and convert biological communities to be tolerant if sensitive species or genotypes are replaced by resistant ones. This phenomenon has been reported in both experiments and field studies with pesticides and also with heavy metals. These changes usually drive to a simplification of the community composition and function and a loss of natural biodiversity.

The potential ecological effects associated with the presence of emerging contaminants in the environment has been largely ignored, and there is a lack of data regarding long-term low-dose exposure

Recent reviews analyzed the potential risk of pharmaceuticals to freshwater biota, and summarized the results of bioassays (e.g. Fent et al. 2006). Pharmaceuticals introduced into the environment may affect animals like they do humans. However, as hundreds of different medical drugs are regularly used, when released into the environment they may interact, producing unknown effects. At least sub-lethal effects have been reported for mixtures of pharmaceuticals at concentrations similar to those found in the environment, pointing to a risk for the biota. For instance, the heart rate and fecundity of the freshwater crustacean *Daphnia magna* are sensitive to beta blockers, the psychiatric drug fluoxetine affects reproduction in the freshwater snail *Physa acuta*, the anti-inflammatory indomethacin produces changes in insect larvae growth, and antidepressants, lipid regulators, steroids, non-steroidal anti-inflammatory drugs, lipid regulators or antibiotics, induce sub-lethal effects on the reproduction, physiology and behaviour of fish (Corcoran et al. 2010). Ultra violet radiation absorbing chemicals (UV-filters) are added to sunscreens and a wide variety of cosmetics, and thus have been detected in freshwater systems. They bioaccumulate in invertebrate, fish and fish-eating birds. To date, quite a lot of information has been generated about the effects of emerging pollutants on single organisms, but community approaches are very scarce (see an example in 5.4, Ginebreda et al. 2010), highlighting the necessity of future investigations. Experimental investigations on communities are difficult because of the long-term studies required and because endpoints are inconspicuous. In field studies it is difficult to predict the effects of emerging pollutants on communities because the effects of these substances on animals are poorly known (e.g. feminization or changes in behavior), and because other stressors often act simultaneously.

A review of the available information on the toxicity and bioaccumulation of alkyphenols and their metabolites reveals toxicity to fish, invertebrates and algae. Effects have been observed on the growth of testicles, alterations in steroid metabolism, disruption of smoltification (internal metabolic process when a fish adapt from freshwater to marine waters, e.g. salmons) and cause

Box 5.4

Case study: Effect of pharmaceuticals on the macroinvertebrate biodiversity in the Llobregat River (NE Spain)

The Llobregat river (NE Spain; see Figure 5.7) is 156.5 km long and covers a catchment area of about 4,948 km^2. Its watershed is heavily populated (3,089,465 inhabitants in 1999), especially in its lower part, which is located in the metropolitan area of Barcelona. Together with its two main tributaries, the Cardener river and the Anoia River, the Llobregat is a paradigm of overexploited Mediterranean river suffering from urban, industrial and agricultural pressures. The river has a mean annual discharge of 693,000,000 m^3 and near 30% is used for drinking water. The Llobregat receives extensive urban and industrial waste water discharges (137,000,000 m^3/year; 92% comes from the waste water treatment plants) that cannot be diluted by its natural flow (0.68-6.5 m^3/s basal flow).

Among other compounds, pharmaceuticals resulting from domestic use are present in relevant quantities in the wastewater effluents discharged into the river. The potential ecotoxicological effect of these pharmaceuticals can be characterised using the so called **hazard quotients** or **hazard indexes** (HQ). For a single compound HQ is defined as the ratio between its environmental concentration to its long term (chronic) toxicity concentration. If more than one compound occurs simultaneously (as is the case), an overall HQ is obtained by summing up all the single HQ's for every compound present. The most relevant contributing compounds to HQ among those analyzed are shown in the attached figure.

Ginebreda et al. (2006), studied the relationship between the HQ's associated with the presence of pharmaceuticals in different points of the Llobregat River basin with the biodiversity of the macroinvertebrates (measured using an appropriate metric, i.e. the Shannon index), which is a measure of the river ecological status. An inverse relation was found (see Figure 5.7) indicating that the presence of pharmaceuticals can be associated with a loss of macroinvertebrate diversity.

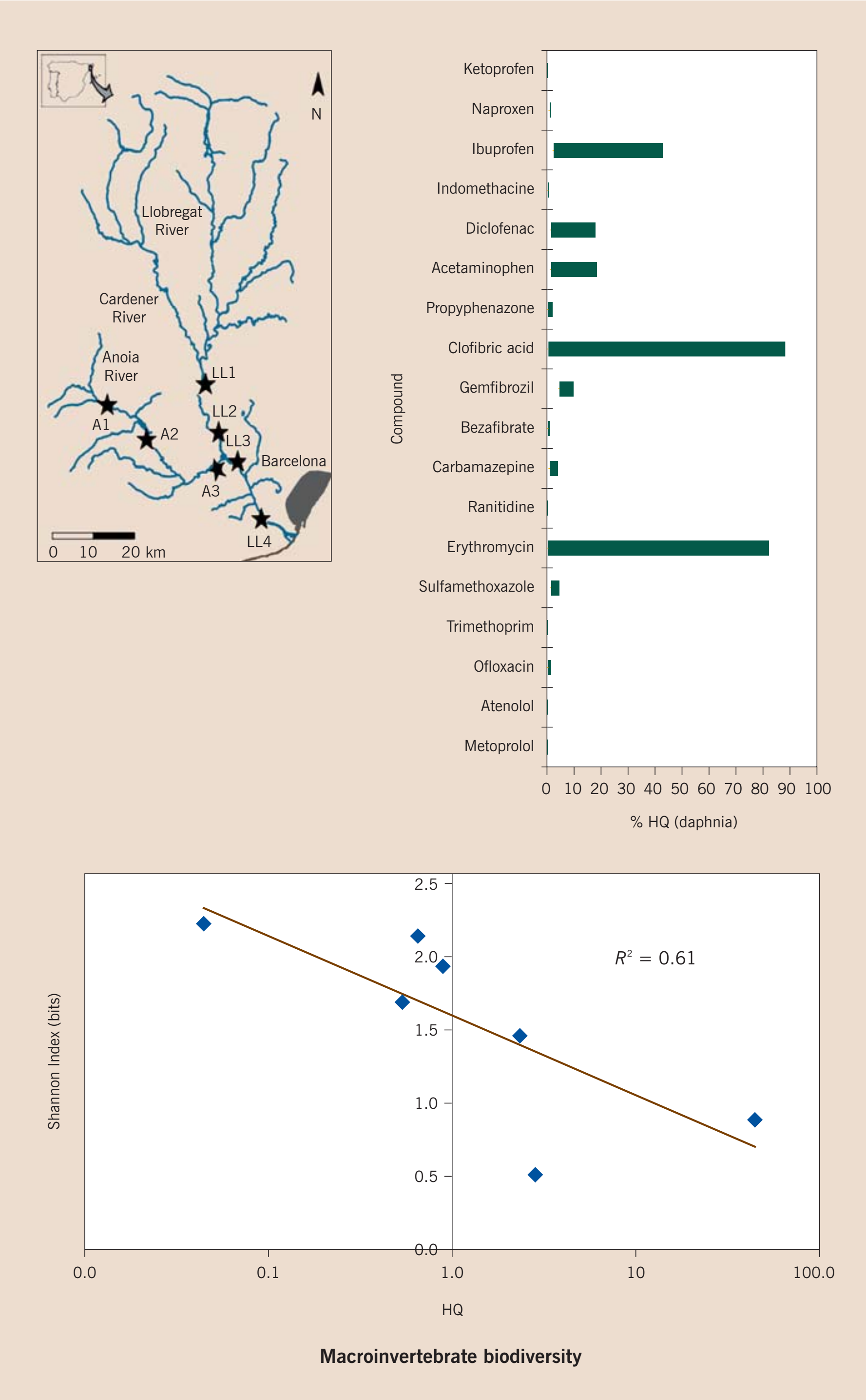


Figure 5.7: *Llobregat River (NE Spain). Relationship between the Risk Quotient (HQ) associated with pharmaceuticals and the aquatic macroinvertebrate biodiversity (expressed as Shannon Index)*

intersex in fish. In rivers, worms and midges are the groups most resistant to alkylphenols. Sub-lethal effects in invertebrates are detected in high concentrations, however, in European rivers, an increase in reproduction of the invasive snail *Potamopyrgus antipodarum* is observed in sediments from sites with high concentrations of xenoestrogens, including alkylphenols. The ability of alkylphenols to bioaccumulate in aquatic biota in the environment is low to moderate. However, high concentrations of perfluorinated compounds are detected in invertebrates, fish, reptiles in aquatic ecosystems, and marine mammals worldwide. More studies have demonstrated the bioaccumulation and biomagnification potential of these compounds in both freshwater and marine food webs. Mortality in sediment dwelling organisms such as the nematode *Chaenorhabtidis elegans* has been observed and decline in fecundity at lower concentrations.

5.5. Present threats and future challenges

Today, one of the major objectives for environmental scientists is to establish causal links between stressors and the quality of ecological systems. The potential ecological effects associated with the presence of emerging contaminants in the aquatic environment have been largely ignored, and there is a lack of data regarding effects on the aquatic ecosystems resulting from long-term low-dose exposure. Such analysis of chronic toxicity of emerging contaminants on organisms is essential to obtain a realistic environmental risk assessment, especially in the case of biologically active pharmaceuticals, because these substances were designed to exert distinct effects. Direct estimation of effects caused by environmental pollutants on ecosystems is not a straightforward task. In real-world scenarios, contaminants rarely exist alone. Instead, they usually appear as mixtures of many compounds, and their combined effects are difficult to predict (i.e. synergies or antagonisms may take place). Furthermore, many other stressing confounding factors (i.e. hydrology, climate change etc.) may take place at the same time, thus giving rise to a very complex situation, especially on Mediterranean rivers. Understanding the respective contribution and reciprocal feed-back of each one on a common integrated picture constitutes a tremendous scientific challenge in which the concurring interdisciplinary efforts of environmental chemists, biologists, hydrologists, engineers and even social scientists are needed.

On the other hand and beyond pure scientific interest, the deployment of adequate management actions at river basin scale as required by the Water framework Directive, such as the elaboration of River Basin Management Plans and the associated Programs of Measures can only succeed if they are supported on a solid scientific basis.

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ECOLOGICAL RESPONSES OF RIVERS UNDER THREAT

Chapter 6

Anthropocene Extinctions: Global Threats to Riverine Biodiversity and the Tragedy of the Freshwater Commons

David Dudgeon

Fresh water is a scarce resource, variously over-used and contaminated, subject to conflicts among humans whose needs are met at the expense of water required to sustain ecosystems. This tragedy of the commons defines the Anthropocene as an epoch marked by river degradation and unparalleled global endangerment of freshwater biodiversity.

6.1. The tragedy of the commons

The story is a familiar one, and has origins in the writings of ecologist Garrett Hardin over 40 years ago. It goes something like this. A villager puts a goat out to graze on the common land around his settlement, so that his family can have a regular supply of milk. Seeing their neighbour enjoying this benefit, each of the other villagers sets their own goat to graze. The village is small, and all goes well until one villager realizes that he can gain more milk by putting out two goats. He does so, and soon his observant neighbours do the same. The numbers of goats increase to the extent that there is less grass for each of them to eat, and thus their per-capita yield of milk is lower than when each villager kept only one goat. The combined yield of the two goats is nonetheless greater than that from a single goat, so the villagers are better off. Soon, one of the villagers is tempted to put a third goat on the commons; his neighbours follow suit. A fourth goat is added… and so on. The additional increment of milk from each goat decreases as the goat population increases, but so long as the villagers obtain some benefit from adding another animal, the number of goats on the commons increases. The additions continue until the grass on the commons can no longer withstand the intensity of livestock grazing. It dies back, the goats starve, and the supply of milk to the villagers dries up. The lesson here is that protection of the environmental commons requires individuals to forego some gain: rather than maximizing the amount of milk they can obtain in the short term, it is wiser to limit the number of goats and optimize the long-term gain of milk by ensuring the commons is not overgrazed and thereby managed in a sustainable fashion.

Why is the tragedy of the commons relevant to fresh water and rivers? Water is an irreplaceable resource for humans and biodiversity, and consumption or contamination of water by one group of human users renders it unavailable or unfit for other users. Furthermore, water is used in a number of ways that are often incompatible: for instance, the extraction of river water by farmers for irrigation makes it unavailable to sustain fish stocks and impacts those who make a living from fishing. Other uses of the same water if it remained in the river channel might include generating hydropower, flushing wastes downstream, allowing navigation, or sustaining biodiversity. Because such uses for humans and non-humans often conflict, fresh water is the common resource *par excellence*. Moreover, equitable use of shared water requires human users to forego gains: the farmer must limit the water he extracts for irrigation so that users downstream can enjoy some benefit; likewise, the industrialist must treat effluent – thereby limiting profits – rather than simply discharging untreated waste water. The tragedy of the freshwater commons is that individual users rarely forego gains voluntarily, yet the rest of the community of users must share the negative consequences of those gains. In short, it is in the interest of individual water users to over-extract or to contaminate because they profit more from doing so than from not doing so; polluters also benefit from the convenient fact that river water flows downhill so their impacts are felt elsewhere.

The potential for conflict among user groups is evident from consideration of the benefits arising from construction of a hydropower dam on a river. People dwelling downstream of the dam, or in cities some distance away, receive the benefits of flood control and electricity. More locally, farmland may be inundated by the reservoir formed behind the dam, and the livelihoods of fishers are compromised by changes to river ecology. In this example, the impacts of the dam are felt locally, typically by the rural poor, whereas the benefits accrue some distance from the site of the dam. All too often, decisions about dam construction are made by city-dwellers who have more political influence than people who are directly affected by the dam and receive no benefit from it. To put it another way, the freedom (or "rights") of parties who stand to gain economically from generating electricity conflicts with the freedom (or "rights") of others to derive livelihoods from the intact river. In any case, scant consideration is given to the need to conserve aquatic biodiversity or preserve ecosystems when conflicting human interests are at stake. An outstanding example of this potential for conflict, and the resulting damage to biodiversity of river fishes, their fishery and human livelihoods along the Mekong River, is shown in Box 6.1. This case has yet to play out fully, and so the possible extent of its implications remains unclear.

Box 6.1

Conflict over the freshwater commons: The case of the Mekong River

The Mekong is an international river that flows through China into the Lao People's Democratic Republic (PDR) and Thailand (where it constitutes part of the boundary between these two countries) thence into Cambodia and Vietnam (Figure 6.1). Its biodiversity has yet to be fully inventoried, but may include as many as 1300 fish species, placing it among the top three rivers in the world in terms of fish richness (Dudgeon 2011). The portion of the river downstream of China, referred to as the Lower Mekong Basin (LMB), supports the world's most productive freshwater fishery, with annual catches (fishes plus shrimps and frogs) amounting to around 2.5 million t worth almost US$4 billion at first sale and perhaps close to twice that as processed products. To put this in context, it represents one quarter of the estimated global freshwater catch. Much of this bounty is based upon a suite of around 50 species of migratory fishes. The importance of this multispecies fishery is evident from the fact that fishing is at least a part-time activity of 40 million inhabitants of the LMB, and the protein obtained from this source is of great dietary significance, especially in Cambodia and land-locked Lao PDR.

The migratory patterns of Mekong fishes are complicated, and different parts of the LMB may support different migratory species that follow a variety of routes at slightly different times. These combine with variations in the topography of the land and extent of the floodplain to result in differing fishery yields across the LMB, with catches being greatest in the lowest section of the river where the floodplain is most extensive (Figure 6.1). As a generalization, migrations of the majority of species are linked to the annual flood cycle, with upstream or lateral movements of fishes initiated by increased flows and floodplain inundation at the start of the monsoon season in May. Migrations are accompanied by breeding, and return movements of adult fishes from upstream or the floodplain – as well as the arrival of young-of-the year – takes place when water levels fall as the monsoon wanes during September or October. Thus seasonally-fluctuating flows, return migrations and floodplain inundation are all essential features of the productive LMB fishery, and the yield from the river depends

Box 6.1 *(cont.)*: Conflict over the freshwater commons: The case of the Mekong River

on sustaining the natural flow pattern and unimpeded movement of fishes.

Conflicts over how best to manage this all-important fishery have been thrown into stark relief by plans of the Lao PDR to build a hydropower dam on the Mekong mainstream (Dudgeon, 2011). The 49 m high Xayaburi dam (Figure 6.1) will have a 100 km long reservoir with a dramatically different flow regime from the river mainstream. It will be a barrier to up- and down-stream migrations of fishes (and downstream transport of drifting larvae), and trap sediments and associated nutrients that would otherwise be transported to downstream portions of the LMB. Since the dam will be situated in the less-productive upper migration zone (Figure 6.1), overall LMB fish yields may be reduced by less than 10% but, within Lao PDR, the reduction in the floodplain fishery could be 70%. Offset against this loss would be the economic gains from the generation and sale of electricity (mostly to neighbouring Thailand), but it seems remarkably short-sighted of the Lao PDR government to trade this off against devastation of a natural larder that provides a significant portion of the nation's animal protein needs. This is indicative of conflict of interests among those making policies and many rural inhabitants likely to be affected by them. The downstream riparian states, particularly Cambodia, are deeply concerned about the possible impacts of the Xayaburi dam on LMB fisheries, and have voiced concerns at the Mekong River Commission (MRC), an inter-governmental organization established

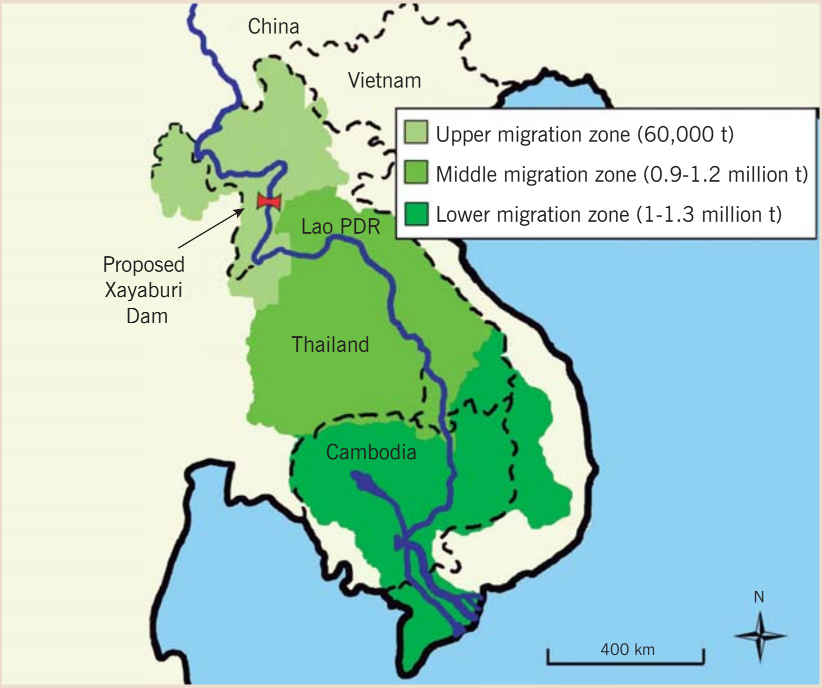

Figure 6.1: *The Lower Mekong Basin showing the annual catches from the three main fish migration zones. The location of the planned Xayaburi Dam within the upper migration zone is also shown*

in 1995 by the four LMB riparian states with the aim of facilitating sustainable development, management and conservation of the river. Despite the need for Lao PDR to obtain the agreement of the other MRC member states to any plan to build a mainstream dam, at the time of writing no consensus has been reached and site preparation has begun. Resolution of this international conflict may be problematic as the MRC has no mandate to interfere with the decisions made at the national level by any of its members. Moreover, unilateral action by the Lao PDR may well result in dam construction by the other nations who will likely see little benefit in continued cooperation via the MRC; indeed, there are draft plans for a further 10 mainstream dams in the LMB. After an environmental assessment of these dams in general, and the Xayaburi dam in particular, the MRC called for a 10 year deferral of any decision of dam construction in the LMB, citing potential livelihood risks for over 2 million people. However, it is not clear whether this appeal will have any effect on the Lao PDR since it conflicts with the perceived national interest. Here, again, the tragedy of the freshwater commons is made evident, as one nation appears intent on pursuing a course of development that will reduce the value of the shared fishery resources of the entire LMB.

Conflicts over multiple uses of water by different stakeholders and interest groups can be difficult or impossible to resolve without mutual compromise. Even when agreement can be reached, only the water which remains after human needs have been satisfied is available to sustain ecosystems. Accordingly, nature often receives a manifestly inadequate share, as demonstrated by instances where flows of some of the world's great rivers (the Colorado, Nile, Indus, Ganges and Yellow rivers) have failed to reach the sea. Some external control must be imposed to ensure that water is no longer treated as a commons. If this is not done, the resource is monopolized by the most powerful human users, leaving little or nothing for weaker parties, or for nature. This, perhaps, is the real tragedy of the freshwater commons.

6.2. A global geography of river threat

A recent global analysis of threats to river health (see Box 6.2) underscores the consequences of conflicts over the freshwater commons, and the consequences of the scant consideration given to biodiversity in explicit or implicit decisions about water-resource management or water allocations (Vörösmarty et al. 2010). The analysis addressed threats to human water security (i.e. a reliable supply of clean water plus protection against floods) and threats to riverine biodiversity separately, since the impacts of a particular stressor will differ greatly depending on whether its effects are felt by river fishes or humans. For

Box 6.2

A global geography of river threat

A recent study by Vörösmarty et al. (2010) set out to map the aggregate effects of a range of threat factors and stressors (termed drivers) on human water security and freshwater biodiversity. The two analyses combined 23 weighted drivers within four categories as set out below to provide a global geography of threats to rivers (Table 6.1).

This list of drivers does not encompass all potential threats or stressors, in part because of the shortage of global datasets at a pixel-scale resolution of 0.5º (i.e. grids of 55.5 x 55.5 km), especially those relating to biotic threats; those concerning physicochemical threat are much better represented. Nonetheless, the range of drivers is wide and, incidentally, indicates the range of threats to rivers and their biodiversity (see also Table 6.2, page 139). Some drivers were routed downstream (if their effects were not inherently local) or divided by annual discharge (if their effects were subject to dilution), and all were weighted according to their relative impacts. The weightings assigned to each driver within each theme, and assigned to each theme, depended on whether their impacts were on biodiversity or on human water security. For instance, the weightings assigned to the number of dams and the extent of river network fragmentation in the context of human water security were quite different from their weightings in calculations of impacts on biodiversity, because dams can benefit humans but are detrimental to riverine biodiversity. Weightings assigned to other drivers that were detrimental for both humans and biodiversity, such as pollutants, also differed between the two analyses since, for example, high loadings of phosphorus and, especially, suspended solids,

Table 6.1: *Threat factors and stressors on human water security and freshwater biodiversity*

Category 1: drainage-basin disturbance	**Category 3: water resource development (i.e. dams and flow regulation)**
— Cropland area — Impervious surfaces — Livestock density — Wetland discontinuity	— Dam density — River fragmentation — Consumptive water loss — Human water stress — Agricultural water stress — Flow disruption
Category 2: pollutants	**Category 4: biotic threats**
— Soil salinization — Nitrogen loading — Phosphorus loading — Mercury deposition — Pesticide loading — Sediment loading — Organic loading — Potential acidification — Thermal alteration	— Number of non-native fish species — Percentage of non-native fish species — Fish pressure — Aquaculture pressure

are relatively more detrimental to biodiversity. In addition, the beneficial impacts of technological advances in engineering and regulatory approaches that enhance human water security were accounted for in order to map "adjusted" human water security; no such adjustment was possible for aggregate threats to biodiversity (for details, see Vörösmarty et al. 2010). Note that these analyses both summarise levels of relative threat to biodiversity and human water security; they do not demonstrate the actual status of human or animal populations as a result of these threats.

As is evident from Figure 6.2, rivers draining large areas of the Earth experience comparable and acute levels of threat. While sources of degradation in most rivers are similar, their engineered amelioration (included in the "adjusted" upper map in Figure 6.2), which emphasize treatment of the symptoms rather than protection of resources, reduces the imposed threat in Europe and North America. However, such technological fixes are either too costly for many other nations or have yet to be adopted. The reliance of some nations on costly technological remedies to safeguard human water security fails to address the underlying threat factors or stressors, and could thus be viewed as a source of water insecurity. In addition, a lack of comparable investments to conserve biodiversity account for the observed declines in freshwater species globally, even in those

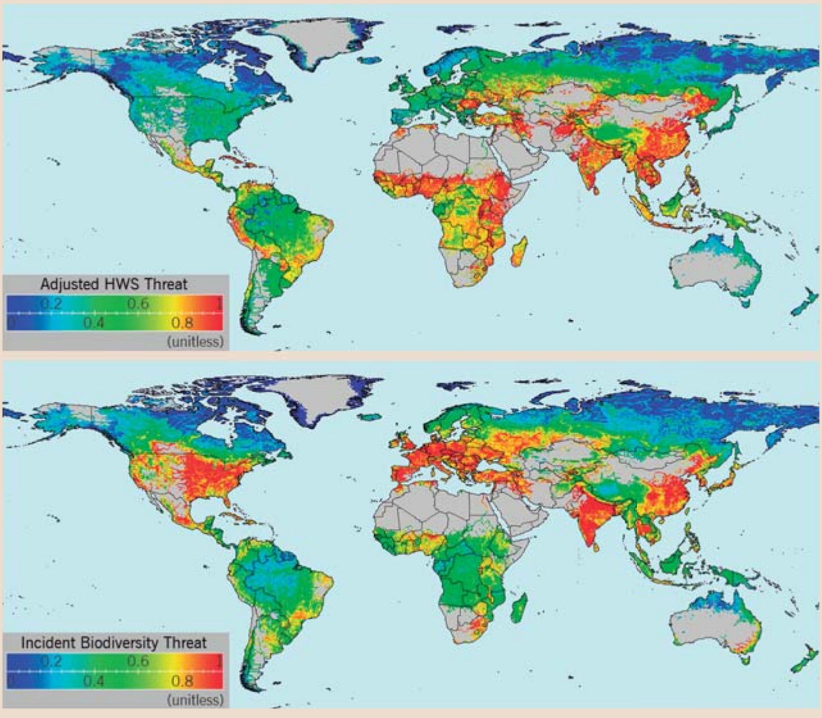

Figure 6.2:
A global geography of river threat, showing the patterns of aggregate threat from a range of factors to – in the upper map – human water security (adjusted to account for investments in infrastructure related to water engineering and treatment) and – in the lower map – freshwater biodiversity. Areas shaded gray have no appreciable river flow

Source: www.riverthreat.net (see also Vörösmarty et al. 2010).

Box 6.2 ***(cont.):*** **A global geography of river threat**

countries where significant adjustments to ensure human water security have been made. Again it must be stressed that the lower map in Figure 6.2 shows only aggregate threats to biodiversity, and not the consequences for populations and species. The best current source of such data are species-level assessments in the IUCN Red List (IUCN, 2011) although, obviously, a comparable analysis showing the aggregate impacts of these 23 drivers would be desirable, and would certainly serve to highlight parlous global plight of riverine biodiversity. A related issue is the need to translate the results of such analyses into action and transformation of current practices of water management: that remains a major challenge.

instance, as mentioned above, the construction of a dam will benefit some human stakeholders and disadvantage others, whereas the effects on fishes – due to altered river flow and habitat conditions, blocked migration routes, and so on – are always detrimental. To give other examples, mercury deposition poses a greater threat to humans who are at the apex of the food chain, than it does to most freshwater plants and animals, whereas acid rain or thermal pollution (arising from water used to cool industrial processes) can have profound impacts on freshwater biodiversity, but negligible effects on humans. This means that the various threat factors must be weighted separately in each analysis according to their relative impacts on human water security or biodiversity.

A surprising outcome of the global geography of river threat is that the two analyses produced similar patterns: low levels of water human security and high endangerment of biodiversity are generally correlated (Box 6.2). However the match between the two is far from complete as Figure 6.3 shows, and there are significant areas of the world, mainly in Europe, North America and Australia, where threats to human water security have been ameliorated (by considerable investment in hard engineering solutions and water treatment) whereas biodiversity remains imperiled: thus conditions are “good” for humans and “bad” for biodiversity. Over much of the rest of the globe, and especially in densely-populated parts of the developing world, the spatial pattern of threats to human water security and biodiversity are remarkably congruent: conditions are “bad” for both humans and biodiversity (Figure 6.3). In places where there are relatively few humans, such as the Amazon, and the far north of Asia, North America and Australia, rivers experience generally low levels of threat (things are “good” for humans and biodiversity) but this state of affairs is increasingly the exception rather than the rule. Most notable, is an absence of places on Earth where human water security is at risk in the absence of any threats to freshwater biodiversity (Figure 6.3). In short, this global analysis reinforces the conclusion that the freshwater commons

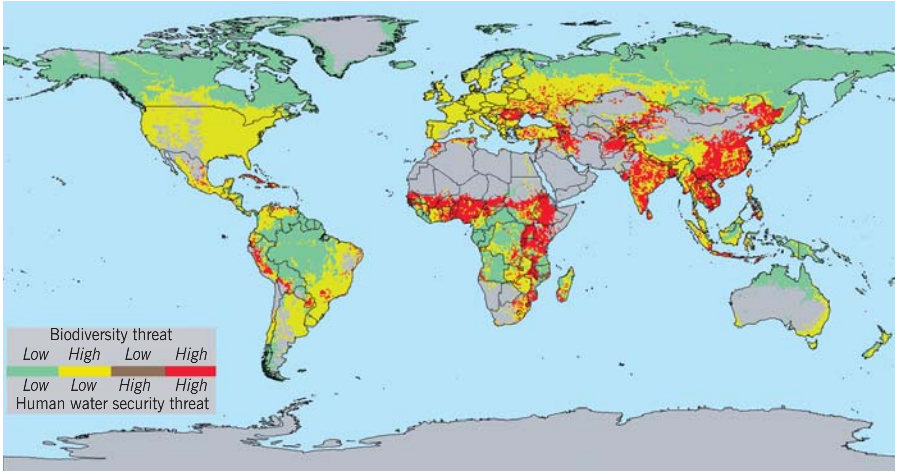

Figure 6.3:
A global geography of river threat, showing the patterns of spatial concordance of aggregate threat from a range of factors (see Box 6.2) to human water security and freshwater biodiversity. Especially striking is the lack of any localities where the threat to human water security is high and that to biodiversity is low. Areas shaded gray have no appreciable river flow

Source: www.riverthreat.net (see also Vörösmarty et al. 2010).

gives rise to state of affairs where human requirements for water invariably trump those of nature. Vörösmarty et al. (2010) do not take any account of the likely consequences of climate change for water availability in rivers, and some of the likely outcomes will be described below. Suffice to say here that climate change projections do not augur well for riverine biota in regions where the human footprint is pervasive, since this is where conflicts over water are likely to be most intense and, thus, the prognosis for biodiversity is especially bleak.

6.3. Principal threats to the freshwater commons

In the Anthropocene world, where many Earth-system processes are dominated by anthropogenic activities (see Chapter 1), we face a "pandemic array" of human transformations of the global water cycle (Alcamo et al. 2008), including changes in physical characteristics, and biogeochemical and biological processes in freshwater systems. These, together with rapid shifts in water use and withdrawal – such as a four-fold increase in demand for water over the last 50 years – are causing dramatic changes in patterns of water stress. The future health and sustainability of river ecosystems will depend upon how humans use water and manage drainage basins. The prognosis is not good. A significant proportion of the Earth's population (~0.9 billion people) does not have ready access to drinking water, and perhaps 40% (>2.5 billion) of people lack adequate sanitation (WHO/UNICEF 2008). The result is a parlous situation where child deaths attributable to contaminated water number around 5,000 *daily* (~1.5 million annually). Thus there is an unarguable imperative to improve access to water and sanitation for millions of people. This, among other things, will drive further transformation of the world's rivers.

The variety and number of threat factors and stressors included in the global geography or river threat study (see Box 6.2) indicate the potential for ecosystem degradation, but they represent a partial list comprising up of only those variables for which data were available at a global scale (Vörösmarty et al. 2010). These and others, such as extraction of river sand for use by the construction industry or the pollution arising from mining activities, have not been captured in that analysis, and nor, as indicated earlier, have the consequences of climate change. A more complete list of the panoply of such threats or stressor is given in Table 6.2. Irrespective of minor differences in the exact nature or relative intensity of threats to individual rivers, the general categories of such threats is fairly uniform the world over, as set out below:

— Flow alteration, water extraction and dam building
— Pollution of many types
— Degradation of floodplains and drainage basins
— Over-exploitation of fishes and other animals
— Invasive species (introduced or non-native organisms, including escapes from aquaculture)
— Climate change

Interactions among these threats or stressors give rise to combined effects that are difficult to predict: for instance, extraction of water for irrigation reduces the diluting effect that rivers can have on pollutants thereby amplifying the impact of the contaminant. Such interactions may be exacerbated by climate change: warmer temperatures and reduced river flows will likely increase the physiological burden of pollution on the aquatic biota, and biological feedback between stressors (e.g. climate change and nutrient pollution) may produce unexpected outcomes. Four of the five threat categories arise directly from the abuse of the freshwater commons since both over-extraction and contamination of water are in the interests of the individual but not the wider community of users. Drainage-basin degradation and habitat destruction are another aspect of the same phenomenon whereby individuals maximize the use of land for cultivation, grazing, timber harvest and so on. The exacerbating factor, in this instance, is that rivers are landscape receivers within drainage basins, and exhibit lateral connectivity with their surroundings. Under the influence of gravity, any increases in soil erosion, nutrient loads and contaminants that accompany land use change (including urbanization) are transported downhill into valley bottoms and hence rivers. Their landscape position not only makes rivers vulnerable to whatever changes occur within

Table 6.2: *Categories of threat to river ecosystems and a summary of their main characteristics and impacts on biodiversity. While this list is not intended to be fully comprehensive, the examples given include most major threats. Other categorizations are possible (see Table 6.4, p. 147): for example pollution, sand-mining and channelization could be grouped together under the shading of "instream habitat degradation", but the categorization is less important than the illustration of the variety of threats that rivers face*

Threat category	Characteristics and examples
Pollution	Broadly defined as something occurring in the wrong place, or at the wrong time, in the wrong (usually excessive) amount — Definition helpful as it avoids stipulating that a pollutant must be an un-natural or man-made contaminant; for example, rivers can be polluted by too much of a naturally-occurring nitrogen compounds, and not just by industrial effluents and toxic chemicals Origins may be "end-of-the-pipe" point sources or more diffuse — For instance, discharge from a factory or a mining operation versus run-off from agricultural land Pollutants may be organic or inorganic compounds, or a mixture thereof — Includes livestock waste and sewage (including pharmaceuticals), discharges from chemical factories or food-processing industries, seepage from landfills, oily runoff from roads and impermeable surfaces, agrochemicals (fertilizers or pesticides), and so on — Combined effects of mixtures of pollutants may be more damaging than individual effects, and have unexpected consequences Can include non-chemical alteration of environments in which pollution is not caused by a substance — Such as cooling water from power stations raising river temperatures (= thermal pollution), or increased suspended sediment loads associated with soil or river bank erosion Direct or indirect effects — Can act directly through toxicity or changes in acidity, causing mortality or sub-lethal fitness reductions, or indirectly by reducing dissolved oxygen levels resulting in respiratory stress
River regulation	Dam construction markedly alters flow conditions to which riverine biota are adapted — Changes flow upstream of dam; impoundment of standing water replaces section of flowing river — Alter flows downstream; natural flow regime replaced by pattern of water release determined by dam operations; in extreme cases, downstream flows may cease entirely for periods as dam (re)fills — Barriers to movement of organisms and material — Physicochemical characteristics of water (dissolved oxygen, temperature, sediment loads) up- and downstream of dam altered Channelization — River flow characteristics altered by channel straightening and constraints of "hard" concretized banks; increased rates of run-off in engineered channel — Levees or barriers prevent exchange of water with – and inundation of – floodplain — In extreme cases, natural habitat entirely destroyed as river channel replaced by channel with concrete sides and base

Table 6.2 *(cont.)*

Threat category	Characteristics and examples
River regulation	Flow reduction due to water abstraction — Over-abstraction of water for irrigation or other human needs reduce flows and, in extreme conditions, may result in dewatering downstream Water transfers between drainage basins — Change flow conditions in contributing and recipient rivers, and may lead to changes in water chemistry of latter; allow exchanges in biota thereby facilitating invasive species
Drainage-basin degradation	Urbanization — Impermeable surfaces dramatically increase magnitude and rates of run-off, and contribute pollutants of many sorts Agriculture — Runoff higher and faster than from natural vegetated land; runoff and groundwater seepage contains agrichemicals and nutrients from fertilizers or animal wastes; soil erodes from farmland during high rainfall events Changes in vegetation cover — Total or partial removal of natural vegetation alter run-off patterns and may be associated with soil erosion and instream sedimentation — Replacement of natural vegetation with different water requirements changes patterns of water supply from soil and run-off; may also alter types and amounts of organic matter (e.g. leaf litter and wood debris) entering rivers, as well as extent of shading and hence, river temperature
Over-exploitation	Reductions of fish stocks — Initially impacts larger or long-lived, late-maturing species, resulting in "fishing down" the food chain and exploitation of smaller, faster maturing species — The use of destructive fishing practices such as poisons, or of electricity and fine-meshed nets, drive further over-exploitation and may be resorted to as large fish become increasingly scarce Reductions of frogs, water snakes, river birds and pearly mussels — Mostly exploited as a source of food, especially in Asia, where the largest freshwater snake "fishery" in the world occurs at Tonlé Sap Lake, Cambodia — Birds that colonially nest in floodplain or riparian forest, or on sand bars in rivers, vulnerable to collection of eggs or nestlings for food — Pearly mussels are exploited for food and formerly also for their nacreous shells and pearls Reductions of crocodiles and turtles — Some exploitation for food, but other valuable products include the hides of crocodiles and shells or flesh of turtles that are used in traditional Chinese medicine; increasing scarcity of target species drives up their value and stimulates further exploitation — Growing prosperity of China has led to import of turtles from all parts of globe (especially other parts of Asia) to supply demand for medicines or tonics

Table 6.2 *(cont.)*

Threat category	Characteristics and examples
Over-exploitation	Collection for global pet trade — May affect some fishes and herpetofauna, as rare or wild-caught specimens can fetch high prices Sand mining — Sand or alluvium is widely-used to make concrete for building, leading to destruction of river habitat
Non-native species	Impacts depend on identity of introduced invader and the receiving community — Carnivorous species especially problematic for native prey with no specific anti-predator adaptations; similar impacts on aquatic plants may also occur if voracious herbivores become established — Competitive interactions for food or space may result from interactions with invasive species — Invasive species may change habitat conditions making them less suitable for native species — Non-native species may introduce diseases to recipient communities — Hybridization may occur if there is a close evolutionary relationship between non-native and native species
Synergistic impacts	Threat factors and stressors will not act in isolation, and their combined effects may be hard to predict, and greater than the sum of their individual impacts — Water abstraction by humans will reduce the capacity of rivers to dilute pollutants — Flow regulation and, for example, pollution change habitat conditions that may favour invasive species; drainage-basin degradation further alters river conditions facilitating invasion — Pollution and habitat degradation may limit ability of populations to recover from or compensate for human exploitation — Overexploitation and population reduction of native species may provide opportunities for establishment of invaders
Climate change	Impacts arising from rising temperatures and long-terms shifts in rainfall patterns, as well as medium-term effects such as glacial melt, and increased frequency of extreme climatic events — Higher temperatures will mean greater water use by plants (crops, pasture and natural vegetation) and thus more water abstraction for irrigation — Conditions in rivers may no longer be favourable for species that evolved there; opportunities for dispersal to suitable habitat may be limited — Human adaptation to a more uncertain climate is likely to encourage dam construction for water storage, flood control and hydropower, thereby magnifying impacts of flow regulation on biodiversity — Altered river flows (increased floods and droughts) will interact with all the threat factors above, while warmer temperatures may increase the toxicity of pollutants, leading to further uncertainty about the severity of their combined impacts

the drainage basin. They are also downstream transmitters of the material they receive so that human impacts do not remain local within a particular section of river. The hierarchical architecture of rivers and their tributaries which ensures that this transmission takes place increases the vulnerability of biodiversity throughout the network.

This longitudinal dimension of river connectivity is also evident from the impacts of dams on habitat conditions downstream. Dams also “smooth out” flow variability and limit floodplain inundation, both of which are essential components of healthy rivers to which the flora and fauna are adapted and upon which their life cycles may depend. Other impacts include the impediments dams cause to migrating fishes, and the entrainment of organic material, sediments and nutrients that sustain habitats and food webs downstream. Dams have led to the elimination of salmon runs in northwest Europe as well as along the west and (especially) eastern coasts of the United States (Limburg and Waldman 2009); the impact is especially severe when it occurs in association with a targeted salmon fishery. Less well known are the impacts on other migratory species, including those that move between rivers and coastal waters (shad, alewives, sturgeon and eels), and the many potamodromous fishes that undertake breeding migrations within river systems such as the Amazon, Mekong (see Box 6.1) and many others. Paradoxically, then, the longitudinal connectivity of rivers that ensures that insults can be transmitted throughout the system – thereby increasing the vulnerability of aquatic biodiversity to human impacts – is a feature essential to ecosystem health, since the migrations of animals and transport of materials depends upon it.

Another manifestation of the tragedy of the freshwater commons is overexploitation of fishes and other animals (mainly turtles, frog and crocodiles) since it is in the short-term interests of the individual to capture yet one more fish now rather than leaving it in the river where it would contribute to the sustainability of the fish stock. Climate change is likewise a consequence of human misuse of the global atmospheric commons, and the inability or unwillingness of individual states (and even individual citizens) to limit carbon emissions. Of the five threat categories or stressors, the effects of invasive species is the only one that does not involve treatment of fresh water as though it were a commons, but it can, nevertheless, interact with threat factors that fall into that category. Disturbed or degraded rivers are more susceptible to invasion by non-native or alien species than intact systems (see also Chapter 8), and they, together with reservoirs and man-made lakes created behind dams, can serve as stepping stones for the spread of invaders to other water bodies. The ongoing global epidemic of dam construction and fragmentation of rivers by impoundments (Nilsson et al. 2005) not only has direct effects on biodiversity through changed flow and habitat conditions, but also facilitates invaders and their impacts on native species by way of predation, competition and so on.

6.4. Understanding the intensity of threats to riverine biodiversity

The global geography of river threat described above is an alarming illustration of the prevalence of human impact on these fresh waters attributable, in large part, to their use as a commons. The range and variety of threat factors or stressors is also noteworthy. But the implications of this state of affairs for humans and biodiversity, and its seriousness, stem from a specific attribute of fresh water, especially water in rivers: its absolute scarcity. As described in Chapter 1, liquid fresh water covers less than 1% of the Earth's surface, and the amount habitable by animals constitutes only 0.03% of the total global water volume and mainly resides in lakes; the amount in rivers and streams is a mere 0.0002% (or 0.006% of all fresh water): a standing volume of 2,120 km^3 (Shiklomanov 1993). This tiny fraction in rivers is the source of most water used by humans.

Estimates of human appropriation vary somewhat, but current withdrawal is slightly over 50% of the accessible surface water supply or "available runoff" of approximately 12,500 km^3 (Chapter 2). Estimates of the proportion withdrawn – e.g. 54% is widely quoted – are sensitive to assumptions about how much of a river or its flow can be regarded as accessible (e.g. rivers in far northern latitudes are mostly untapped), or available for capture (typically floodwaters are not), and to the magnitude of total global annual runoff (probably ~40,000 km^3). Given that the Earth's population has recently topped 7 billion, and can be projected to reach 9 billion by 2050 or thereabouts, the intensity of competition for water between humans and nature must inevitably increase, raising concerns that planetary boundaries for sustainable use of this resource may be overstepped in the foreseeable future (Rockström et al. 2009). Such competition for water is always highly asymmetric: as human requirements for water go up, that which remains for nature declines; the converse is *never* true.

One driver of competition, among others (see above), will be the demand for water to grow food for the additional humans. Agriculture already accounts for roughly 70% of water withdrawals and, while only around 15% of global croplands are irrigated, they yield half of the saleable crops. Given that the extent of arable land is finite (and limited), bringing a greater proportion under irrigation may be the most expedient approach to feeding the 2 billion additional people expected by 2050, and improving the nutritional status of the many who are presently undernourished. This will further diminish the volume of water remaining for nature, a situation that will be exacerbated by shifts towards diets incorporating more animal protein because approximately twice as much water is needed to produce an American diet than a vegetarian diet of equivalent calories. One estimate is that food security needs could result in of water for irrigation consumption increasing by up to 50% over the next 20 years (Rockström et al. 2009).

To make matters worse, fresh water is not only a scarce resource: fresh waters are also hotspots of biodiversity. Approximately 125,000 freshwater species have been described and named by scientists; they represent 9.5% of known animal species on Earth, including around one third (over 18,000 species) of all vertebrates (Dudgeon et al. 2006; Balian et al. 2008). The latter are mainly fishes, but also comprise the entire global complement of crocodilians, virtually all of the amphibians, and most of the turtles. Many of these are semiaquatic, and include species confined to riparian zones or adjacent floodplains (Dudgeon et al. 2006). Moreover, despite the much greater area and total production of marine environments, fish (Actinopterygii) species richness in the seas and fresh water is similar (14,736 and 15,149 respectively), with all of the saltwater species derived from a freshwater ancestor (Carrete and Wiens 2012). Some freshwater vertebrates are, of course, associated with lakes rather than rivers. Nonetheless, the fact that almost 10% of the Earth's animal biodiversity is associated with a relatively tiny amount of fresh water covering less than 1% of the planet's surface, stands in stark juxtaposition to ever-growing human demands for water which sustains that diversity. Indeed, Marshall McLuhan's catchphrase "the medium is the message" serves as uncomplicated summary of the essential threat to riverine biodiversity.

A further complicating factor is that most freshwater species have limited dispersal abilities, and their habitats are aquatic "islands" set within a terrestrial matrix. Fish typically are unable to move between rivers since they cannot tolerate salinity sufficiently well to migrate along the coast nor can they travel overland and surmount terrestrial barriers between drainage basins. Amphibiotic animals, such as frogs and aquatic insects, which have aquatic juveniles and terrestrial adults, enjoy more scope for dispersal over land. However, mayflies, caddisflies and most other stream insects (with the exception of some dragonflies) are weak fliers or habitat specialists, as are many amphibians, and their ability to traverse the terrestrial landscape is limited. Because of the limited faunal exchange between river basins, and the insular nature of inland waters, there is a considerable degree of local endemism (high α-diversity) and the inhabitants often have small geographic ranges, resulting in high species turnover (β-diversity) among river basins. Effective barriers to dispersal may explain the relative richness (in per unit-are terms) of fishes in freshwater habitats (Carrete and Wiens 2012), and have an important implication for biodiversity conservation. Individual river basins (especially those in latitudes unaffected by recent glaciation) are often not "substitutable" in biodiversity terms, and thus protection of one river does not ensure preservation of a representative portion of the regional species total (γ-diversity). To put it another way, loss of a species from a single river could, in effect, represent global extinction. This is markedly different from the relatively localized effects of most human impacts in terrestrial landscapes. Because rivers serve as receivers and transmitters of human impacts, are insular, and have drainage networks with a hierarchical structure, insults from upstream can travel throughout the system with the potential to imperil aquatic animals downstream.

6.5. The next great extinction?

Freshwater biodiversity is in a state of global crisis with freshwater species generally far more imperiled than their terrestrial counterparts (see reviews by Dudgeon et al. 2006; Strayer and Dudgeon 2010). Population trend data compiled by WWF since 1970 indicate that declines in freshwater species are considerably greater than those on land (Loh et al. 2005), especially in the tropics (WWF 2010), and the IUCN Red List (www.redlist.org) reveals that a host of freshwater species are extinct or imperiled (Table 6.3).

Table 6.3: *Threatened freshwater animal species and, where relevant (in parentheses), their terrestrial (reptiles, mammals) or (fishes, decapods) marine counterparts, as indicated by an analysis of the IUCN Red List (version 2011.2). Data are percentage of extinct, critically endangered, endangered or vulnerable species out of the total number assessed. The proportion of species classified as data deficient is shown also. Marine bivalves have not been included, as only 30 species have been subject to IUCN assessment*

	Fishes	Frogs	Reptiles	Mammals	Decapods	Bivalves	Dragonflies
Number assessed	5,719 (2,912)	5,609	338 (3,226)	145 (5,404)	1,864 (250)	428	2,654
Threatened species (%)	30 (7)	30	37 (24)	40 (22)	19 (0.4)	38	10
Data deficient (%)	18 (20)	26	11 (17)	13 (15)	40 (35)	17	30

Note: Fishes = Actinopterygii; Frogs = Anura; Decapods = crayfish, freshwater crabs and shrimps.

A recent analysis argued that human activities have transgressed planetary boundaries for terrestrial and marine biodiversity, with species losses at least one to two orders of magnitude in excess of background extinction rates derived from the fossil record (Rockström et al. 2009). Assuming this is correct, we must also have far exceeded whatever margins would have been sustainable for freshwater biodiversity. Moreover, inadequate knowledge of tropical freshwater biodiversity (Balian et al. 2008) – especially among invertebrates – means that the extent of threat may be even greater. For example, fully 30% of all species of frogs and toads are at risk, but another 26% of these animals are classified by the IUCN as data deficient (DD), indicating that there is insufficient information on their distribution and abundance to make a reliable conservation assessment. In some such cases, it is very likely that an absence of records may well represent records of absence, and those DD species are likely to be gravely endangered. Significantly, the DD categorization in the Red List carries the caveat that if the range of a species is circumscribed and a considerable period has elapsed since it was last recorded, threatened status may well be justified. Among other groups of freshwater animals (Table 6.3), reptiles, mammals and decapods include more threatened species than their terrestrial or marine counterparts, and decapods and dragonflies include a high proportion of DD species. Assessments for many animals groups are far from complete: for instance, only 30% of fishes and 35% of reptiles have been assessed. Nonetheless, a striking finding is that almost 50% of freshwater animals assessed (20,524 species) by the IUCN are threatened (25%) or data deficient (23%); the equivalent total number for terrestrial animals (30,340 species assessed) is 36% (23% threatened and 13 % DD); for marine (6,414 assessed) it is 27% (14% threatened and 23% DD).

In intensively-developed regions, often those where the global geography of river threat reveals that human requirements for water have been secured by investment in river engineering and water treatment, over one third of the species in some major groups are threatened, including 38% of the fish species in Europe and 39% in North America. Other notable examples of species declines (reviewed by Dudgeon et al. 2006) are large river fishes worldwide, Asian freshwater turtles, and the recent extinction of the Yangtze river dolphin, *Lipotes vexillifer*. To obtain an overview of the threats to riverine biodiversity, and compare them with the threats facing freshwater animal biodiversity in general as well as species in other realms, data included in the IUCN Red List (version 2011.2) can be analysed to determine the percentage of extinct, critically endangered, endangered or vulnerable species at risk from a particular threat factor. As Table 6.4 shows, species are generally threatened by two or more factors acting in combination (2.7 on average for riverine animals) with biological resource use (or overexploitation) comprising the major threat overall, and in rivers also. However, pollution (Chapter 5) is of almost equal importance as a threat to biodiversity in rivers, and is the major threat to freshwater animals in general, but is less important in other realms, especially on land. Agriculture and natural system modification are also important threats to rivers, as are commercial development and invasive species, with climate change currently perceived to be a less important threat to freshwater animals than to their marine counterparts (especially coral). While there are more threatened terrestrial species than freshwater species (and some of the terrestrial animals may be better be characterised as semiaquatic), the numbers of threatened species in fresh water, and

Threat categories	Rivers (freshwater)	Terrestrial	Marine	All realms
Biological resource use	18 (17)	23	24	21
Agriculture	15 (14)	24	3	18
Urban development	11 (11)	12	13	11
Invasive species and pests	11 (11)	11	14	11
Pollution	17 (18)	5	14	11
Natural system modification	12 (13)	8	3	10
Climate change	6 (6)	6	15	7
No. threatened species	2,893 (5,206)	7,022	897	11,150
Mean no. threats per species	2.7 (2.1)	2.3	3.0	2.1

Table 6.4: *Relative intensity of threats to biodiversity in rivers, and in marine and terrestrial realms, as indicated by an analysis of the IUCN Red List. Threats to species across all realms are shown also, as well as (in parentheses) threats to freshwater species in general (i.e. all inland waters). Data are percentage of extinct, critically endangered, endangered or vulnerable species at risk from a particular factor. Factors that threaten fewer than 5% of species in any realm (i.e. human intrusion and mining, both 4% across all realms; all other factors combined <1%) have not been included*

especially in rivers, is high relative to the area these habitats occupy: there are only around 2.5 times more threatened animal species in the terrestrial realm than in rivers, or 1.4 times more than in fresh waters as a whole. This finding is likely to be relatively robust as there is no reason to suppose that assessments of the conservation status of terrestrial animals are any less complete than those of freshwater species and, as mentioned above, almost half of all freshwater species assessed are either at risk of extinction or DD. Some threatened species are associated with two realms (e.g. salmon or sturgeon that migrate between rivers and the sea) and are represented twice in the calculations in Table 6.4, but these are unlikely to have influenced the outcome of the analysis of relative threat intensity. However, the threat categorization used by the IUCN notably affects the conclusions that can be drawn from Table 6.4, as it is both less detailed and more generalized than one applying to rivers alone (Table 6.2). For instance, logging is treated by the IUCN as a category of biological resource use, whereas dams and flow regulation represent modifications of natural systems, but are categorised separately from threats due to agriculture (including plantations, livestock rearing and aquaculture) or urban development which could both be considered as alteration of natural habitats. Since rivers are markedly affected by land use changes within their drainage basins, as well as in-stream modifications such as flow regulation, the IUCN categorization does not fully capture the variety and intensity of threats to biodiversity in rivers set out in Table 6.2.

Irrespective of the causes of species declines and losses, they will certainly have knock-on effects for other organisms: for instance, reductions in predatory species may "release" smaller prey from control allowing them to proliferate; conversely, reductions in prey species will have implications for the animals that feed on them. For instance, birds, bats and spiders that make use of riparian zones can be impacted by changes in river water quality that reduce the survival of aquatic insect larvae and hence the abundance of emerging adults that sustain terrestrial insectivores. Other impacts are also possible: depletion or annihilation of salmon runs by overfishing and dam construction sever the connection between the sea and headwater tributaries by way of which marine-derived nutrients are transported upstream by migrating salmon. The result is reduced productivity of streams and associated riparian forest because of the absence of nutrients that would normally be contributed by death and decomposition of the breeding salmon. Terrestrial species such as bears that feed upon migrating salmon can be affected also (see Chapter 10).

It may well be possible that loss of significant portions of riverine biodiversity will represent the first wave of the sixth mass extinction event in geological history that eminent biologists believe is now ongoing as a result of human transformation of the Earth system (Eldredge 2001 http://www.actionbioscience.org/newfrontiers/eldredge2.html). The extent of the declines and losses of freshwater biodiversity that have been documented is probably a reliable indicator of the extent to which current practices are unsustainable (Dudgeon et al. 2006), and demonstrate how human exploitation and impairment of rivers have outpaced our best attempts at management. To this can be added a substantial extinction debt (that is presently impossible to quantify) due to human actions that have been taken already that have reduced populations below levels from which they can recover (Strayer and Dudgeon 2009), as well as losses that occurred in the past that have been overlooked. One likely source of this debt is habitat fragmentation (e.g. by dams), which interacts with the insular nature of rivers and their geometry (see above), to reduce the viability and persistence of populations that may already be dwindling to extinction.

As the Anthropocene Epoch proceeds, trajectories of human population growth, water use and consequential environmental alterations are rising steeply (the "great acceleration"; see Chapter 1) and can be projected to continue in the near future, likely resulting in further extinctions and knock-on effects, placing riverine biodiversity under greater stress.

6.6. Imperiled river invertebrates: The pearly mussels

One group of animals that particularly well illustrates the vulnerability of freshwater fauna to an array of anthropogenic threats is the pearly mussels. These bivalve molluscs (part of the group consisting of mussels and clams) make up the order Unionioda, consisting of around 850 species in six families, the majority of which are placed in the Unionidae (Figure 6.4). All threatened freshwater bivalves (see Table 6.3) are pearly mussels, and 8% of them (32 species) are already extinct. Their vulnerability arises from their own inherent attributes, as well as their interactions with other species.

Pearly mussels are especially diverse in large rivers in China and in those parts of the United States that escaped glaciation during the Ice Age, but occur also in Southeast Asia and elsewhere; the tropical species are especially poorly known. Many species have confined distributions and a high degree of endemism compared to other invertebrates such as dragonflies and other aquatic insects that can disperse during the terrestrial adult stage. Even in relation to fully-aquatic animals such as fish, mussels are relatively immobile or sedentary. A restricted range is one of the main contributors to vulnerability of freshwater species

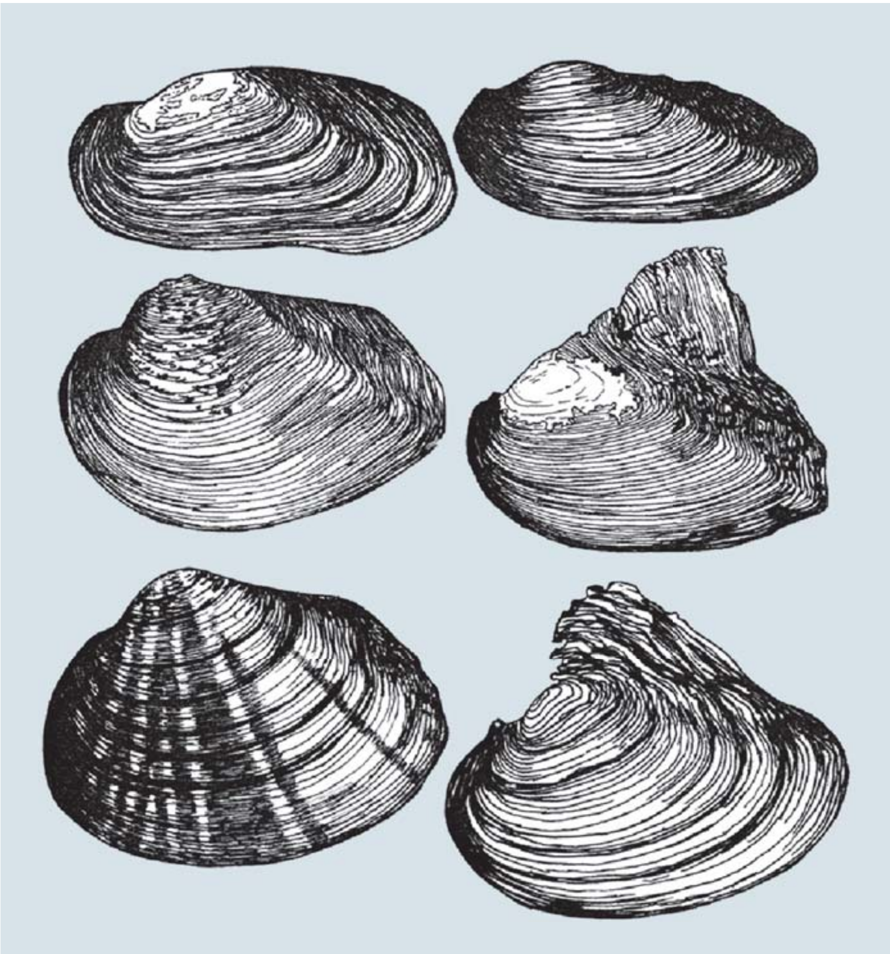

Figure 6.4:
Typical representatives of different pearly mussel (Unionoida) genera from China. The shell length of adults can vary from 3 cm up to almost 25 cm

since local degradation of their habitat can cause the loss of a population and may even result in global extirpation. Moreover, because they are filter feeders and burrow in sand and gravel of river beds, pearly mussels are acutely sensitive to water quality and sedimentation resulting from the wash-off of soil and silt from agricultural land. Flow modification or channelization that affect patterns of riverbed erosion and deposition also reduce habitat suitability for unionids.

Unusually for a freshwater invertebrate, many unionid populations have been depleted by human exploitation, driven by demand for their nacreous shells (Figure 6.5), the quest for pearls, or consumption of their flesh. In some parts of the world unionids are an important subsistence food, and rarer species may be taken as by-catch even if more abundant species are targeted. In the United States, mussels formed the basis of a substantial pearl industry beginning in the 1850s; around 10 species were involved, and as pearls were present in as few as one mussel in a 1,000, there was much mortality for little gain. Eventually, populations became overexploited and insufficient to sustain the industry, which collapsed in the 1900s (Humphries and Winemiller 2006). Beginning in 1890, a wider variety of mussels were collected and their shells used for button manufacture, but less than 20 years later, many larger species had declined and attention had shifted to smaller species. Some harvests seem astounding: in 1913 alone,

Figure 6.5: *An example of a large (24 cm long) unionid from Thailand showing the nacreous ("pearly") interior* (up) *shell valve and the worn exterior* (down) *valve*

over 13 million kg of shells were removed from living mussels in Illinois, and 100 million mussels were taken from a single 73 ha bed in the Mississippi River (Strayer 2006). Over-expoitation devastated the mussel fauna to such an extent that they have yet to recover, and the loss of a substantial biomass of filter-feeders must have had a significant impact on food webs and transport or transformation of suspended organic matter, phytoplankton and so on. While mussels no longer experience high levels of exploitation in the United States (due in part to the replacement of mussel-shell buttons by plastic substitutes in the mid-20th century), they drove the historic decline of many species so that – as with large river fishes – recollections of mussel abundance are subject to baseline shift (Humphries and Winemiller 2006). In parts of the lower Yangtze basin in China, however, a pearl "industry" continues, based on culture of a few relatively hardy species (mainly *Hyriopsis cumingii*, but *Cristaria plicata* and *Sinandonota woodiana* have been used) yielding virtually all of the global supply of freshwater pearls.

To make matters worse, since 1985, mussels in the eastern United States have suffered from competition with the non-native and highly invasive zebra mussel which has a relatively short life cycle and rapid growth *(Dreissena polymorpha)* leading to the extirpation of many populations, and this process may be driving already-threatened species to extinction. In this case the competition for filtered food is aggravated by the tendency of zebra mussels to foul or overgrow the unionids by attaching themselves to the shell of the larger mussels and the aggregate filtration rate of the attached individuals may greatly exceed that of their hapless host (Figure 6.6).

Figure 6.6:
Zebra mussels (Dreissena polymorpha) *overgrowing the posterior, siphon-bearing end of* Lampsilis siliquoidea *the United States. These invasive non-native mussels attach to pearly mussels and compete with them for food*

The multiple threats facing unionids have had the consequence that around 70% of the approximately 300 species of unionid in the United States are federally classified as in danger of extinction, with more than 10% perhaps already extinct due to human activities. Equivalent assessments from China have not been undertaken, but anecdotal reports suggest widespread declines in unionids. In addition to the impacts of pollution, habitat degradation, overexploitation and invasive competitors, one specific attribute of unionids places them at further risk: their life cycle. For most mussels and clams, the majority of which are sea-dwelling, reproduction is a simple matter. Eggs and sperm are released into the water, where they meet and fertilization takes place leading to a planktonic larval stage. Larvae feed on planktonic algae and develop until they are ready to metamorphose into a benthic juvenile. This lifestyle is not well suited for river-dwelling pearly mussels, in part because river currents might sweep planktonic larvae downstream and out to sea, and in part because river water contain much less algal food that the surface waters of the sea. Instead, pearly mussels depend on the presence of a suitable host to complete their life cycle. Females incubate fertilized eggs in modified gills (termed marsupia) where they develop into larvae called glochidia. The glochidia are expelled into the surrounding water and attach to the fins, gills or skin of a fish host (a few may also attach to amphibians or turtles). There, the glochidia live as parasites for several days or weeks (sometimes longer), whereupon they metamorphose into a tiny mussel and drop off the host to become free living on the river bed. While some unionids seem to rely on little more than chance, and the production of prodigious numbers of larvae, to locate a host, in others the margins of the flesh protruding from the shell of gravid females serve to attract potential hosts. Simple adaptations involve the use of contrasting colours along the tissue margins of gravid female mussels, but the lures may be expanded and elaborated (Figure 6.7) to resemble the shape and markings of a small fish bearing, in some *Lampsilis* species, a distinct eye spot. The resemblance is further enhanced if the lure sways in the current. The function of such lures is to attract the attention of other fishes in search of a mate or a meal, thereby greatly increasing the chances that larvae expelled at an appropriate moment, or released when the fish strikes at the lure, will locate a host. A similar system is used by *Villosa iris* but, in this case, the tissue margins are highly elaborated to resemble a small crayfish (Barnhart 2008). In the genus *Ptychobranchus* the glochidia larvae are released in groups encased within an ovisac (Figure 6.8A). The posterior end of the ovisac is adhesive, attaching to cobbles or stones, while the anterior portion waves to-and-fro in the current. Depending on mussel species, the ovisacs resemble potential prey items such as insects or larval fish. When an unsuspecting fish bites, the ovisac ruptures (Figure 6.8B) to release a cloud of glochidia (Figure 6.8C) that attach to

Figure 6.7:
The fish lure of Lampsilis cardium *mussels feature marked striping of the tissue margins, reminiscent of the markings of some species of North American* Notropis minnows *(Cyprinidae)*

the host's gills (Figure 6.8D). Among other adaptations to attract fishes is provision of edible clumps of sterile eggs to serve as a "bribe" for potential hosts (Barnhart 2008).

Figure 6.8:
A) Glochidia of Ptychobranchus subtentum *mussels are enclosed within 2 cm long ovisacs that resemble aquatic insects - in this case, blackfly pupae (Simuliidae). B) Glochidia are released in clouds when the ovisac is ruptured by, for instance, a fish bite. C) Glochidia (each ~0.2 mm long) prior to host attachment. D) Glochidia attached to the gills of a fish host*

This array of adaptations illustrate the essential point that pearly mussels cannot complete their life cycles in the absence of an appropriate host. And not just any fish host will do; many do not provide favourable conditions for glochidial development, while species-specific variations in glochidium morphology permit attachment to some types of hosts but not others. This presumably explains why mussels vary in the form and appearance of their adaptations to lure fish, and it is tempting to suggest that the more derived or highly evolved the lure appears then the more specific the most-parasite relationship can become. While there is not always a one-to-one relationship between fish host and unionid, the majority of pearly mussels depend on a few hosts only; in extreme cases the match between fish and mussel can be specific to a particular drainage basin. In any event, the parasitic larval stages of each type of mussel are constrained to a greater or lesser extent by the variety of available hosts. Reproductive failure need not involve complete disappearance of the preferred hosts: once encounters between glochidia and hosts fail to exceed some critical threshold, the probability of successful larval encystment, or the chance that a metamorphosed juvenile will drop from its host into a habitable patch of riverbed habitat, become too low to support recruitment of the next generation. While the ecological requirements of tiny mussel juveniles are not well understood, they are certain to differ substantially from those of the adults (e.g. with respect to sediment grain-size). Mussel vulnerability to anthropogenic impacts is consequently increased because habitat for populations of these animals must meet the requirements of all life stages.

Unionids are therefore directly imperiled by an array of factors, in addition to the indirect threats posed by others (e.g. dam construction, overfishing) that affect the distribution and abundance of their hosts. The more specific the host-parasite relationship, the more likely it is that impacts on the fish will be detrimental to the mussel. Despite their curious interactions with fishes (see also Box 6.3), and their species richness, it is difficult to bring conservation attention to bear on unionids since, like many freshwater invertebrates, they are non-charismatic – many species look quite similar, especially when the fish lures are not evident (Figure 6.4) – and have little contemporary relevance for most people.

6.7. Shifting baselines

Our imperfect knowledge of past conditions in rivers gives rise to “shifting baseline syndrome”. Most of the factors that threaten freshwater biodiversity today also acted in the past, although their scale and intensity has increased recently. Fish and other aquatic animals, such as beaver *(Castor canadensis),*

Pearly mussels and potential fish extinctions

Box 6.3

A further complication in the relationship between fishes and pearly mussels arises from the fact that not only do the mussels parasitize fish and depend upon them, but the mussels are themselves an important link in the life cycle of certain fishes. Small carp-like Asian and European fishes known as bitterlings (~30 species, mainly in the genera *Acheilognathus and Rhodeus,* within the cyprinid subfamily Acheilognathinae) depend upon unionids as an egg repository, and are unable to reproduce in their absence. During the breeding season, females develop a long thin ovipositor that can be inserted between the shell valves in order to deposit eggs on the gills (Figure 6.9). The male releases sperm in the immediate proximity of the mussel and the sperm are carried into the shell and onto the gills – where the eggs are fertilized – by the feeding currents. The bitterling eggs and larvae are protected from predators as they develop, and compete with their filter-feeding host for food and oxygen. Free-swimming juveniles escape from the shell at three to four weeks of age. At least two species of bitterling are considered vulnerable by the IUCN. The plight of Japanese *Rhodeus smithii,* classified as critically endangered, is manifestly more serious, and Chinese *Acheilognathus elongatus* may even be extinct. Their decline has been attributed mainly to extirpation of potential hosts, although pollution and competition with introduced species are implicated also. Undoubtedly, bitterling dependence on pearly mussels for breeding makes them more susceptible to anthropogenic impacts than most other fishes. The plight of these fishes may provide a basis for building a compelling case for conservation of pearly mussels since the knock-on effects of mussel loss seem certain to include extinction of bitterlings.

Figure 6.9: *Male* (left) *and female* (right) *European bitterling* (Rodheus amarus) *showing the ovipositor used to deposit eggs within the unionid host*

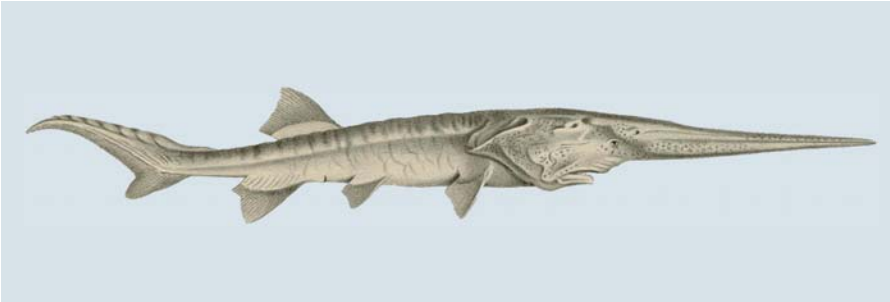

Figure 6.10: *The Yangtze paddlefish,* Psephurus gladius, *one of only two species in the family Polyodontidae. It is classified by the IUCN as critically endangered, but may already be extinct. Adults were reputed to grow to 7 m in length*

Figure 6.11: *The critically-endangered Yangtze sturgeon,* Acipenser dabryanus, *also known as Dabry's sturgeon, is confined to parts of the Yangtze upstream of the Three Gorges dam and appears close to extinction. It attains no more than 20 kg, much smaller than the Chinese sturgeon,* Acipenser sinensis *– also critically endangered – which occurs in the lower Yangtze and may weigh up to 450 kg and exceed 3 m in length*

have experienced historical declines since mediaeval times (around 1000 AD) in Europe, caused by a combination of siltation from intensive agriculture, increased nutrient loads and pollution, proliferation of mill dams, introduction of exotic species, over-fishing and hunting beaver (Hoffmann 2005). In the 17th and 18th centuries, these impacts were exported as migrating Europeans exploited those parts of the world that had hitherto been influenced only by indigenous peoples. Because these impacts occurred well before any stock formal assessments, they give rise to the false impression that conditions in the immediate past (or at the point when a human observer first begins to take an interest) reflect conditions in the intermediate and distant past: i.e. deception and a tendency to underestimate the extent of human impacts due to a shifting baseline (Humphries and Winemiller 2009). The shifting baseline is not just a matter of historic interest: large and charismatic species exploited by fishers can be affected by baseline shift within the span of a human generation; when these species are not encountered on a fairly regular basis, they are rapidly forgotten. This breakdown in expectation of what species should be present in rivers, and thus what needs to be conserved or restored, has been dubbed "ecosocial anomie" (Limburg and Waldman 2009). This point has been well demonstrated along the Yangtze river (Turvey et al. 2010) site of the recent extinction of the Yangtze river dolphin, and where extensive surveys have failed to detect any Yangtze paddlefish (*Psephurus gladius*: Figure 6.10) – the world's longest freshwater fish – or Yangtze sturgeon (*Acipenser dabryanus*: Figure 6.11) – and where the Chinese sturgeon (*Acipenser sinensis*) has become vanishingly rare. These large and charismatic species were rapidly forgotten by local communities as soon as they failed to be encountered on a fairly regular basis, offering a striking example of rapid cultural baseline shift within a human generation.

One implication of shifting baseline syndrome is that if people cannot remember what has been lost, or what conditions were formerly like in rivers, then it becomes difficult to manage these ecosystems in ways that will allow the recovery of already rare or threatened species. At the same time, target conditions for restoration of degraded systems have been forgotten. Furthermore, restoration of rivers back to their pristine state is no longer practical given the all-prevailing human footprint on most landscapes. Instead, it may be more realistic to plan for river rehabilitation where management is directed towards enhancing native biodiversity – that is, improving conditions relative to current baselines – rather than attempting to achieve a restoration goal that may prove impractical or unfeasible, prohibitively expensive and hence not societally acceptable. An example of rehabilitation of a riverine species, albeit one that was more an outcome of serendipity than advance planning, is given in Box 6.4. It illustrates the opportunities than may remain for conservation of near-extinct species that have been long forgotten by local communities.

Back from the Brink

Box 6.4

Père David's deer or milu (*Elaphurus davidianus*) is – or was – an inhabitant of swampy river floodplains in central and southern China (Figure 6.12). Milu are amphibious and strong swimmers, spending considerable time in the water as well as on grasslands and in reed beds; their hooves, resembling those of cows, are adapted to soft ground, and they graze a mixture of grasses and aquatic plants. Because of the productivity of the floodplain habitat, milu can reach 200 kg and are larger than the majority of terrestrial deer, and the males have large and many-branched antlers. Milu numbers were reduced, especially during the last 1,000 years or so, by habitat loss (due to conversion of floodplain to rice paddy) and hunting. By 200 years ago, they were approaching extinction, and the last wild individual was shot in 1939 (Jiang and Harris 2008).

Milu became known to western science in the 1860s through the observations

Box 6.4 *(cont.)*: Back from the Brink

of missionary Père Armand David. By that time, almost all of the remaining animals were part of a herd that had been maintained in the Royal Hunting Garden, by a succession of Emperors, for over 500 hundred years. A few of these milu were subsequently transported to Europe, which proved fortunate since a series of accidents and political upheavals in the late 19th and early 20th century resulted in the complete destruction of the imperial herd. Subsequent survival of milu was due to maintenance and captive breeding of descendants of the exported animals at Woburn Abbey in England. Although classified as extinct in the wild by the IUCN (Jiang and Harris 2008), milu from the English herd were sent to China in 1985, where there is now a substantial number of captive animals. Of greater importance is that two wild populations have since been successfully established along the Yangtze: the first at Dafeng Reserve (Jiangsu Province) in 1986 and, later, in 1993 at Tianezhou Reserve (Hubei Province). Both "reintroduced" populations have expanded considerably, and limits to the quantity of habitat set aside for them have led to incursions of milu from reserves into surrounding farm land. There are constraints upon how much habitat remains in which milu can range freely, but for now, it seems that this large riverine species has been rescued from the brink of extinction. Memories of this deer would have long-since disappeared as a result of cultural baseline shift along the Yangtze, and milu serve as a good example of how attempts at riverine restoration could be misled by overreliance upon recollections of what the ecosystem was once like.

Figure 6.12: Elaphurus davidianus

6.8. Climate change

Species loss from rivers may have been overlooked because baseline shift causes us to underestimate the extent of ecosystem degradation. The future seems likely to hold even more species loss from inland waters, as temperature, rainfall and runoff patterns alter as a result of global climate change. A point that will not have escaped readers is that such change is occurring precisely because humans have treated the Earth's atmosphere as a global commons, with individual nations unwilling to restrain their carbon emissions for the global good.

Human-caused climate change represents a profound and insidious threat to freshwater biodiversity (Table 6.2, p. 139), and thus it deserves special attention here. Signs of global climate change in freshwater ecosystems include detection of a direct carbon dioxide signal in continental river runoff records (Gedney et al. 2006), as well as warmer water temperatures, shorter periods of ice cover, and changes in the geographic ranges or seasonality of freshwater animals in temperate or higher latitudes (reviewed by Hein et al. 2009). Current projections are that temperature increases in the tropics will be less than those further from the equator, but the impacts of any rises in lower latitudes could be considerable since tropical *cold-blooded'* animals such as fish, amphibians, invertebrates and so on may already be close to their upper tolerance limits. There is an inverse relationship between temperature during growth and body size in amphibians and many aquatic invertebrates that results in smaller size at metamorphosis, plus decreased body mass due to increased metabolism at higher temperatures, and their combined effects reduce adult fitness. Shifts in the timing of fish breeding and migration (driven by alterations in temperature and/or flow and inundation patterns) are also likely, and warmer conditions could have serious consequences for reptiles such as turtles and crocodiles in which the sex ratio is determined by the temperature of the environment. Potential sources of physical disturbance and stress on riverine species include increased scouring and washout associated with snow melt and flood events, saline intrusion caused by sea-level rise in coastal areas, and the fact that the concentration of oxygen dissolved in water declines as temperature rises. Warmer temperatures and greater water use by terrestrial plants (and the need for more water for irrigation) may mean that some rivers that flowed year-round become intermittent. Climate change may pose further hazard by facilitating the establishment of alien species that threaten native biodiversity, and magnifying the toxic effects of some pollutants.

Because there has been insufficient research on the implications of climate change for freshwater biodiversity, especially in the tropics, the potential for adaptation to warmer temperatures is unknown for the vast majority of species. The best we can do is make extrapolations from the studies of temperate species (especially "cold-blooded" animals that have temperature-sensitive metabolic rates), which may allow identification of "winners" – species that may thrive under the changed conditions – and "losers" – those that fail to adjust and perish. Such extrapolation could, however, prove misleading for tropical species if, as pointed out above, they are already close to their upper tolerance limits. One prediction that seems likely to be robust is that the species most vulnerable to climate change will be those that are highly specialized, with complex life histories, restricted ranges or limited distributions, or highly-specific habitat requirements. The pearly mussels discussed above have most, if not all, of these attributes, and are sure to be placed at further risk by climate change. They will be climate-change "losers".

Freshwater biodiversity is extremely vulnerable to human impacts because almost 10% of the species known to science are concentrated in less than 1% of the Earth's surface area

Climate-change "winners" will be species that are generalist in their habits and habitat requirements, and have short generation times that will increase the possibility of rapid adaptation to changed conditions. But there may be other options allowing persistence. If, for the purposes of simplification, we assume that climate change only affects median water temperature of rivers, one option for species that lack the evolutionary capacity to adapt to rising temperatures (or cannot do so quickly enough) is to shift their distribution. For instance, animals in rivers could, conceivably, compensate for rising water temperatures by moving upstream to higher – and cooler – elevations or latitudes. This could be especially important for species in the tropics that are already close to their upper thermal tolerances and might be feasible for (say) fishes in north-to-south flowing rivers, although such movements would be subject to limitations imposed by river topography, the presence of dams or other in-stream barriers, availability of suitable habitats upstream, or some combination of these. However, the extent of movements needed to compensate for the upper bounds of the range of temperature rises predicted for the next century seem insurmountable for most freshwater species (see, for example, Bickford et al. 2010).

Given the insular nature of freshwater habitats, adaptation to rising temperatures by way of compensatory movements into cooler habitats further from the equator or to higher altitudes are often not possible, especially for the many fully-aquatic species that cannot move through the terrestrial landscape. Furthermore compensatory movements north or south are not possible where drainage basins are oriented east-west. Even flying insects and amphibians than can travel over land might find their dispersal opportunities limited in human-dominated environments. One conservation initiative that could help address this problem would be translocation or aided migration of threatened species from warming water bodies to habitats within their thermal range (Olden et al. 2011). Such actions would be controversial and costly, requiring detailed information about the species (currently available for only a tiny fraction of freshwater species imperilled by climate change), and pose the risk of ecological outcomes of the type associated with introduction of species to locations outside their natural geographic range. The argument that we should not move animals around so as to avoid causing unanticipated harm cannot be equated with adopting the "precautionary principle" because climatic shifts as the world warms may leave freshwater animals stranded within water bodies where temperatures exceed those to which they are adapted or to which they can adjust. Under these circumstances, doing nothing could result in more harm than that the potential risks associated with translocation.

In addition to the direct effects of climate change on freshwater biodiversity, human responses to such change could give rise to indirect impacts on biodiversity that will be as strong or even greater. Climate change will create or exacerbate water-supply shortages and threaten human life and property that will encourage hard-path engineering solutions to mitigate these problems (Palmer et al. 2008), including new dams, dredging, levees, and water diversions to enhance water security for people and agriculture and provide protection from floods so altering flow and inundation patterns in ways that will not augur well for biodiversity. In addition, there is increasing impetus to install new hydropower facilities along rivers to reduce dependence on fossil fuels and meet growing global energy needs. These engineering responses will magnify the direct impacts of climate change because they limit the natural resilience of ecosystems: for instance, by restricting the ability of animals to make compensatory movements to cooler conditions. A related problem is that hard-path solutions initiated in response to disasters (e.g. severe floods associated with rainfall extremes) may be permitted to circumvent environmental reviews and regulations because of the urgent need for project implementation. Offsetting some of the effects of dams will require that their operation be adjusted to ensure allocation of sufficient water to sustain ecosystems and biodiversity downstream. The need for implementation of these environmental flows is already pressing: one estimate is that dams retain over 10,000 km^3 of water, the equivalent of five times the volume of the Earth's rivers; the associated reservoirs trap 25% of the total sediment load that formerly reached the oceans (Vörösmarty and Sahagian 2000). This has had important consequences for rates of aggradation of deltas around the world, causing them to "sink" relative to sea levels and allowing upstream intrusion of salt water (Syvitski et al. 2009). This will exacerbate the effects of sea-level rise induced by climate warming and the consequences for freshwater animals in the lower course of rivers are unlikely to be favourable.

6.9. What is needed?

What can be done alleviate the tragedy of the freshwater commons? Or avert further damage and species declines? An obvious starting point is the necessity to raise awareness – at a variety of levels, from children to policy makers – of the remarkable richness of riverine biodiversity. To this must be coupled the many threats that these organisms face, and – as a consequence – the degree of endangerment that prevails. This primary task can be approached in a number of ways, but will require that we marshal sound arguments for that protection. It is one thing to enlighten people about the hidden or overlooked biodiversity of inland waters, and the extraordinary adaptations some of these animals have evolved (as in the case of pearly mussels, for example), but quite another to mount persuasive arguments for their protection. A fundamental aspect of the tragedy of the freshwater commons is that individuals must limit their own actions so as to maintain the communal good. In the Anthropocene world where conflicts over water are pervasive and likely to grow, limitations upon human activities intended to preserve biodiversity, and justifications of allocations of water for nature, will need to be extraordinarily persuasive. And, to reiterate, "the medium is the message": because fresh water is more limiting than the supply of land nor subject to comparable patterns of consumption and use, and because freshwater animals have far more restricted distributions than their terrestrial (or marine) counterparts, the conflicts between humans and biodiversity are exacerbated. How, then, can progress be made?

Two options seem possible, but these are not mutually exclusive, and other alternatives need not be ruled out. First, the argument for preservation of freshwater biodiversity can be made on utilitarian grounds: i.e. preservation of biodiversity is worthwhile for humans – hence we should limit our selfish degradation of the commons – because of the goods and services that more-or-less intact ecosystems offer. This point has been touched upon in Chapter 1: it suffers from the shortcoming that it is by no means evident that the services provided by river ecosystems (e.g. provision of clean water, flood control, and so on) require preservation of all the organisms present in those systems. There might be redundancy, such that certain species have no unique (or even apparent) function, and thus their loss can be substituted by others (Chapter 7). It might be argued that the supply of ecosystem goods, such as the yield of protein from capture fisheries, may be enhanced by maintaining rivers in near-natural states with intact food chains. This rationale has been (and is being) used in attempts to limit dam construction along the mainstream of the Mekong River (see Box 6.1, and further discussion by Dudgeon 2011) where there is a highly-productive fishery based on exploitation of a large number of species. But not all rivers sustain economically-valuable fisheries, or the fishery may be based on one or a few species. In such cases, managing the river for other uses (e.g. some combination of water supply, navigation, hydropower, and even waste disposal), or in a manner that favours productivity of the most desirable fishery species, may maximize net economic benefit even if it fails to bring about the best overall outcome for biodiversity.

Population declines in freshwater animals, and the proportion of species threatened with extinction, are far greater than their counterparts in the marine and terrestrial realms

One major obstacle to implementation of conservation measures for rivers is that scientists have yet to demonstrate convincingly that there is a strong linear relationship between biodiversity and ecosystem functioning – and hence the goods and services enjoyed by humans. This failure weakens any argument that all species must be conserved if ecosystem functioning is to be maintained (Dudgeon 2010; a detailed account of this matter is given in Chapter 7). In most cases, this failure deprives the conservation biologist of a utilitarian justification for the preservation of many elements of biodiversity or the protection of an intact ecosystem, although the potential detriment to the world's most productive freshwater fishery in the Lower Mekong Basin seems to be a possible exception to this generalization. The only remaining option, therefore, is to assert that freshwater biodiversity deserves preservation, in and of itself, because of its existence value. Such a stance arises from an ethical imperative and comprehension of the shared evolutionary history of all life on Earth. It could also be taken to encompass the inter-generational value that biodiversity could have for our descendants, to which could be added its option value in the broadest sense: i.e. direct uses that certain species may have for humans in future, or contributions – thus far unappreciated – made to ecosystem functioning. Unfortunately, many might argue that none of this offers sufficiently strong justification for prioritizing freshwater biodiversity conservation in light of human needs for clean water and sanitation, nor will it serve to satisfy the expectations of growing populations who wish to enjoy improved standards of living.

Clearly, better communication and raising awareness will be necessary to avert further degradation of the freshwater commons, but this alone will be insufficient. To advance the utilitarian argument for conservation (but not the ethical case), we need compelling evidence of, firstly, a positive relationship between biodiversity and ecosystem functioning. Secondly, the connection between freshwater ecosystem functioning and enhanced provision of goods and services for humans needs to be elaborated. The latter is needed because a utilitarian argument for preserving freshwater biodiversity so as to maintain ecosystem functioning depends on the notion that impaired function does, in fact, reduce the benefits gained by humans. The scientific priority is clear: elucidation of the links between biodiversity, ecosystem functioning, and the consequential benefits to be derived by humans.

Even in the absence of such information, there is much to be done. More research is needed to develop regionally-relevant hydro-ecological models that underpin allocations of water for nature; these environmental requirements must then be compared to the water needed to produce goods and services for society (Alcamo et al. 2008). That will allow identification of regions where conflicts between humans and biodiversity for scarce water resources will be most intense, and where conservation and management challenges should be addressed urgently. Much research on environmental flow allocations in rivers has already been undertaken, and scientists have a good understanding that maintaining the dynamic and variable nature of river discharge is a prerequisite for protecting freshwater biodiversity. This presents a formidable challenge given the context of a resource management paradigm aimed at controlling hydrological variability and enhancing predictability for humans, as well as the need to strike a balance between resource protection and development. Implementation at appropriate scales will be challenging also, but there have been some successes with modification of the operation of small dams to enhance downstream flow conditions. New and innovative strategies to develop regionally-specific environmental water allocations are being researched, and a new framework for flow standards developed by Poff et al. (2010) is evidence of recent progress.

Pearly mussels are threatened by reductions in water and sediment quality, historical overexploitation, and by reliance on the presence of certain fish species to complete their life cycle

More must be done to develop action plans for the conservation of those species that have been categorised as threatened by the IUCN. Such plans would need to incorporate population and/or habitat management, and identify measures needed to protect the target species, as well as regular monitoring. Attention also needs to be paid to data deficient species and their conservation status updated so that they can either be confirmed as currently non-endangered or, alternatively, become the subject of a targeted action plan. In addition to action plans, work is needed to determine which species are most vulnerable to climate change, and might therefore warrant conservation intervention, such as assisted translocation. At present, potential climate-change losers cannot be identified due to the paucity of ecological data on many freshwater species and their thermal tolerances. On a larger scale, it might be possible to identify the rivers that are most likely to be affected by climate change, such as those that are presently fed by glaciers, but taking measures that will alleviate the worst effects of such changes will be challenging as the failure to regulate global greenhouse gases emission to the atmosphere plainly demonstrates.

Another research topic where more work is needed is identification of variables for monitoring riverine biodiversity. The best variables would accurately represent the current status of biodiversity (or, at least, a subset of particular interest; e.g. fishes), and respond rapidly to environmental change; ease of measurement would also be a desirable attribute. Surrogate variables that indicate river health and hence are likely to be correlated with biodiversity may also be useful. Examples might direct measurements of water quality, in addition to some of variables listed in Box 6.2 (Table 6.1, p. 134), but such surrogates cannot fully substitute for direct monitoring of species richness and population sizes of species of particular conservation interest or societal relevance. Without long-term monitoring data, we will be in no position to ascertain whether and in what direction changes in river flora and fauna are taking place. Nor will we be able to assess the success or otherwise of measures to mitigate anthropogenic impacts, or management efforts to rehabilitate or restore river ecosystems. Furthermore, without a clear understanding of the relationship between biodiversity and ecosystem functioning (see above), it is unclear whether the effects of anthropogenic changes in rivers will be first manifest by structural alterations (shifts in species diversity and abundance) or by ecosystem functioning (productivity, nutrient dynamics, organic matter processing, and so on). Research into this topic, and the relationship between biodiversity and ecosystem functioning in general, is needed urgently.

Irrespective of the short-comings of current knowledge or inadequacies of research efforts, one thing is certain. Those interested in conservation of riverine biodiversity must take every opportunity to communicate what is known now about the threats to and endangerment of biodiversity, the value (however it is defined) such biodiversity has for humans, and the steps that can be taken to ameliorate, reduce or remove threat factors. Preservation of river ecosystems and the biodiversity they sustain will require the combined efforts of scientists, managers, politicians and other citizens. It is therefore essential that scientists share their findings, however incomplete they might be, so that they can be acted upon or implemented. The book you are now reading was written with that spirit in mind, and in the hope of ensuring that the Anthropocene epoch is not marked by a mass extinction of freshwater animals and the loss of many natural wonders.

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Chapter 7

So What? Implications of Loss of Biodiversity for Ecosystem Functioning

Sylvain Dolédec and Núria Bonada

A loss of biodiversity is expected to affect ecosystem properties. In most river ecosystems, the relationship between biodiversity and ecosystem functioning relies on a cascade of effects among species identity, sequence of species loss and environmental context. The resulting ecological complexity calls for applying the cautionary principle in conservation and restoration planning.

7.1. What is the problem – or why should we measure the effects of biodiversity loss on ecosystem functioning?

We live in an extraordinary rich and diverse planet, with global estimations of biodiversity ranging from 5 to 100 millions of species, from which only about 1.9 millions of species are known. However, many human activities are leading to a global biodiversity loss at a rate that is higher than what should be naturally expected. This global trend is especially worrying in freshwater ecosystems (Figure 7.1).

For instance, the Living Planet Index (LPI) has been tracking population trends of over 2,500 vertebrate species since 1970 in order to calculate a yearly average rate of changes for those populations. LPI reflects somewhat the health of

Figure 7.1: *A typical Mediterranean river. Mediterranean climate regions all over the world host high number of terrestrial and aquatic species and are considered biodiversity hotspots. Nevertheless, many Mediterranean rivers are currently threatened by rising water consumption, and face bleak prospects as a consequence of ongoing climate change*

planet ecosystems (terrestrial, freshwater, marine). Examining LPI trends shows that the abundance of more than 300 freshwater vertebrate species declined by ~55% from 1970 to 2000, while those of terrestrial and marine systems declined by ~32%. As a result, the scientific and public awareness of the ecological consequences of such a dramatic species extinction has much increased in the last two decades, as well as the budget allocated to conserve and restore biodiversity. For example, during the period 1988 and 2008 the World Bank invested more than US$6 billions to support biodiversity conservation programs. Such programs were not only meant to preserve our natural heritage but also aimed at studying the ecosystem consequences of such biodiversity loss.

The biodiversity of our every day environment results from various ecological processes. First of all, any region cannot accommodate all the potential species on the earth. For example, the platypus is only found in Australia, the hippopotamus lives only in Africa. In European waters, *Echinogammarus berilloni* (Catta 1878), an amphipod (Figure 7.2), is native of the Iberian Peninsula, but the creation of canals connecting waterways has made the species spread and establish in several French rivers beyond its native distribution (see chapter 8 for detailed mechanisms of invasion).

This means that biodiversity differs from one region to another and what we observe today is the result of 3.5 billions years of evolution and various colonisation and settlement processes. Similarly, a locality, for example a stream reach in a particular region cannot accommodate all the potential species of the re-

Figure 7.2:
A male of Echinogammarus berilloni, *an endemic crustacean amphipod of the Iberian Peninsula that lives and feeds on leaf litter of streams*

gion and only those able to pass through environmental filters that characterize the locality will be found (Figure 7.3).

In other terms, a species must possess traits (Box 7.1) that allow it to cope with several environmental constraints operating at a regional scale (essentially climate, relief, geology) and at a local scale (current velocity, temperature and so on). Dispersal ability is a further trait of the species that enables it to colonize a given locality (Figure 7.3).

Finally, in a given site, the individuals of a species have to locally find an array of biotic and abiotic conditions that allow them to survive and reproduce. However, because the resources available in the locality are not infinite, species have to share them with others, which inevitably induce competition among them. As a result, the limited number of potential habitats will ultimately limit the number of species in the site (Figure 7.3). The amount of resources in a given site thus determines the level of biodiversity. Once species are settled, by growing and reproducing they contribute to the functioning of the ecosystem.

Ecosystem functioning refers to a complex group of several functions that sustain the ecosystems. These functions include biomass production, decom-

Box 7.1

Trait, disturbance and habitat templet

Species trait

Any morphological, physiological or behavioural characteristic that characterise the life history of a species, such as body mass, the number of offspring produced each year, the type of respiration, or the type of food ingested. Every environmental change that affects the growth, survival or fecundity of individuals, or their behaviour, will affect the population size through changing birth and mortality rates. Such modification will in turn, affect the arrangement of the community, and ultimately ecosystem functioning.

Species trait state

A particular value or modality taken by a trait, which may vary along environmental conditions and temporally (e.g. small and large are trait states of body mass).

Functional trait

Characteristics of an organism that determine its effects on a given function (e.g. shredders contribute to leaf-litter decomposition).

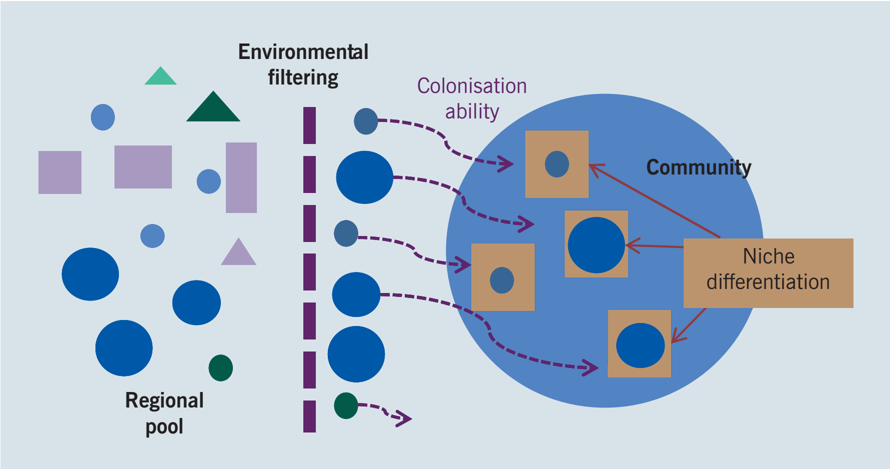
Disturbance

Unexpected event that impacts the community at a local scale and over a short time. Disturbances are defined by their intensity (including magnitude and duration), predictability and frequency. For example, floods are natural disturbances that may change stream morphology and reshape gravel bars.

Habitat templet theory

A theory assuming that habitat provides a spatio-temporal framework on which evolution forges characteristic life histories. The habitat framework adapted for rivers is built along two axes (Townsend & Hildrew 1994). The X-axis represents temporal variability (the intensity of disturbance), the Y-axis spatial heterogeneity. In streams, heterogeneity may mean refuge for organisms and a higher disturbance will have less impact in a mosaic of habitats than if the habitat is homogeneous. To survive in these habitats, species have to possess specific biological characteristics. For example, if a species inhabits a frequently disturbed habitat, then the species can either resist if it has clinging facilities or escape and return easily once the disturbance ceases. In frequently disturbed habitats, species will have short life duration and produce many offspring to ensure survival in a constrained environment. In contrast, when the intensity of disturbance is smaller, a higher diversity of species including those resistant and non-resistant and those resilient and non-resilient may occur in the community.

position of organic matter, or nutrient uptake (see Box 7.2). Many researchers have suggested that ecosystem functioning depends on biodiversity, which could be framed under the biodiversity-ecosystem functioning hypothesis (hereafter called B-EF). This has emerged as a central ecological question by the turning of the century (Loreau et al. 2001). It basically assumes that biodiversity affects ecosystem properties and, therefore, the benefits we obtain from them (ecosys-

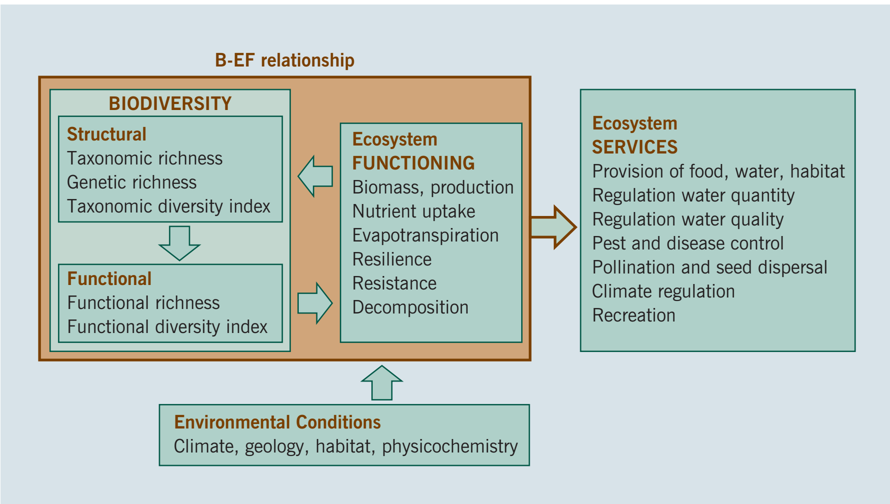


Source: Redrawn from Boulangeat (2012).

Figure 7.3: *Environmental filtering or how individuals of given species can be found in a given stream reach. The regional pool comprises all species that exist in a region. From this pool, some species are filtered out by environmental characteristics (for instance, habitat features), which are unsuitable for them to survive. The species that can survive in the conditions of this reach must also have colonisation ability to reach the locality, either actively (flying, swimming) or passively (water or wind transportation). From the species that reached the locality, only those with ecological niches different enough will be able to coexist, the rest will be eliminated by competition. Environmental filtering and niche differentiation are essential processes for explaining local biodiversity*

tem services) (Figure 7.4). Species in a community perform many ecosystem properties, such as biomass production and mineralization of the organic matter. Therefore, understanding the B-EF relationship may greatly help in understanding the consequences of the current decline in biodiversity in ecosystems.

But, how general is the B-EF hypothesis? The B-EF hypothesis is considered a “long-standing paradigm in ecology” (Caliman et al. 2010) supported by much



Source: Redrawn from Díaz et al. (2006).

Figure 7.4: *Relationship between biodiversity, ecosystem function and ecosystem services (see Box 7.2) and how environmental conditions can modify these patterns*

Box 7.2

Biodiversity, function, functioning and services

Biodiversity

Biological diversity refers to the extent of genetic, taxonomic and ecological diversity over given spatial and temporal scales. Biodiversity includes structural and functional aspects. It can be measured using variables such as richness or diversity indices but species composition may be even more informative about how species are arranged in the assemblage (their relative abundance). **Structural diversity** refers to taxonomic units (species) whereas **functional diversity** refers to the role of these units in the ecosystem (such as feeding strategies or body size).

Function

Ecosystem functions stand for ecosystem processes. They result from the interactions among biotic and abiotic elements of the ecosystem. The term is generally employed to refer to both ecosystem properties and services. Ecosystem properties include stock of energy and materials (for example biomass), fluxes of energy or material processing, (for example productivity, decomposition) (Lecerf & Richardson 2010)

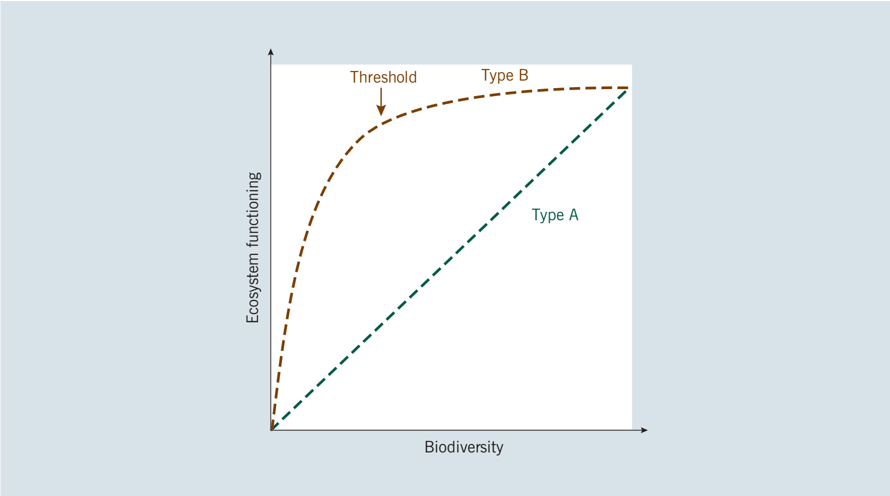
Functioning

Functioning refers to the joint effects of all functions (processes) that sustain an ecosystem. Thus, it considers the combination of biomass, production, decomposition, and nutrient uptake, among other ecosystem characteristics.

Services

Humans can benefit from the different functions that ecosystems provide. These benefits are known as **ecosystem services** and include characteristics grouped as provisioning, regulating, supporting and cultural services.

evidence, which suggests that biodiversity can positively enhance ecosystem functioning. In a literature survey, carried out with 100 studies dealing with the B-EF hypothesis, Srivastava and Vellend (2005) found that a B-EF relationship was significantly positive in 71% of the studies for at least one ecosystem function. Thus, in these cases, species-rich communities will have more efficient ecosystem functioning than species-poor communities. However, very often this relationship is only linear up to a certain point, and adding more species has no effect on the ecosystem functioning. In this case, we consider that the new added species are functionally redundant to the already existing ones. Thus, two general types of B-EF responses can be expected in ecological systems: a linear response where each species is functionally singular and contribute steadily to the ecosystem functioning (type A, Figure 7.5) and a non-linear response where ecosystem function is effectively maximized by a relatively low proportion of the total diversity (type B, Figure 7.5). In that case, few abundant species most implied in the ecosystem function live together with rare species



Source: Redrawn from Schwarz et al. (2000).

Figure 7.5: *Hypothetical relationships between biodiversity and ecosystem functioning. Type A (green) assumes a gradual decrease of ecosystem functioning with successive biodiversity loss events. Type B (brown) supposes functional redundancy (see Box 7.3) since ecosystem functioning is not immediately impacted by biodiversity loss events. The threshold on type B curve indicates the biodiversity limit below which additional species loss will involve a significant reduction of ecosystem functioning*

that have a more minor contribution to ecosystem functioning. Now, when one species disappears, its function can be compensated by the increased abundance of another already existing species. As a result, ecosystem functioning does not immediately decline with biodiversity loss. Many ecological situations probably lay in-between these two extremes. Nevertheless, of the 100 studies mentioned above, only 39% showed a linear relationship (type A, Figure 7.5) whereas 61% showed a non-linear one (type B, Figure 7.5) (Srivastava and Vellend 2005).

Clearly, the two types of responses have different implications in terms of conservation. In type A, at each species loss event, ecosystem functioning decreases in a steadily way. In type B, several species may be functionally redundant (see Box 7.3) and species loss does not change ecosystem functioning beyond a certain value of biodiversity (threshold in Figure 7.5), below which further species loss events may greatly impact ecosystem functioning. Therefore, in the latter case, many species could be lost before detecting any changes in the system because many species are functionally redundant. In addition, the value of biodiversity, or that of particular species, would be very low up to a certain point above which conservation measures would not be justified, if only ecosystem services are considered. However, we have to bear in mind that the presence of a certain type of B-EF relationship depends on the type of ecosystem under study, the type and number of ecosystem functions measured, the range of biodiversity under focus, the type of biodiversity measure used, and the species identities (Ghilarov 2000; Srivastava and Vellend 2005).

Box 7.3

Functional redundancy: A reality or a myth?

Inherent to communities, functional redundancy implies that different species perform the same role in ecosystems, so that changes in species diversity should not affect ecosystem functioning. It is thus assumed that biodiversity is more sensitive to disturbance than ecosystem functioning.

The reality. Considering a single or few ecosystem functions as surrogates of functioning, species having similar functional roles can be considered redundant. For example, when looking at the processing of the matter and energy in a system, all primary producers contribute to the same function (irrespective of their relative contribution). Therefore, studies focused on a single or few functions are more susceptible to find species redundancy (Rosenfeld 2002).

The myth. In a broad sense, all functional roles that a species can perform could be seen as its functional niche. The different traits (see Box 7.1) of a species might respond to different ecosystem functions. For example, feeding traits are related to the processing of the matter and energy in a system whereas body form can be related to resistance. Therefore, local biodiversity can be only explained because species have few overlaps of their functional niche and contribute all together to the overall ecosystem functioning. Simply speaking, if "ecosystem functioning" means all compounds that plants and animals in a community have in their bodies or release in the environment, then any redundancy is impossible (Ghilarov 2000).

The reality or the myth of functional redundancy is related to how ecosystem functioning is defined, either as a single function (the reality, since current studies hardly look at more than two functions, like for example litter decomposition and algal growth in streams) or the multiple functions of an ecosystem (the myth because nobody is currently able to measure all the functions of an ecosystem).

Another important issue in considering the effects of biodiversity loss on ecosystem functioning is that species loss is never random. Some species are more prone to extinction than others for a specific pressure. For example, for regions where climate change will result in longer drought periods, those species not adapted to survive droughts will experience a higher extinction risk. Overall, type B response (shown in Figure 7.5) will occur if those species that are more susceptible to be lost contribute less to the ecosystem functioning. As a result, in addition to accounting for biodiversity, the assessment of the B-EF relationship requires addressing the functional characteristics of species (their functional role, see Box 7.2) as well as their biological traits (see Box 7.1) in relation to their extinction risk and the ecosystem function under focus. Therefore, it is difficult to forecast the impacts of environmental changes on ecosystem functioning. Moreover, it might be risky to consider a single or a few ecological functions as surrogates of global ecological functioning, as the different species may be important for different ecosystem functions.

Nature conservation is usually carried out at a regional scale, and thus humans have established protection areas over hundreds of square kilometres in natural parks. However, studies looking at the B-EF relationship have been assessed at a local scale (in a prairie, in a stream reach, or in a laboratory experiment), which cannot be directly translated to regional scales.

At the local scale, ecosystem functioning can be enhanced with the presence of one or more species because species may have a complementary action, which enhances ecosystem functioning. This action, which is called complementarity of resource use, occurs when part of the local habitat is occupied, namely there remains empty space for other species. In other terms, under the level of total biodiversity that a given site can support (shown by the vertical bar on Figure 7.6), ecosystem functioning is improved by the complementarity among species and increases with biodiversity. Above this level of “optimal” local biodiversity (vertical line), species that are added from other regional localities involve a more severe local competition. Such competition yields a global reduction of their performance and ultimately involves a local decrease of ecosystem functioning (beyond the vertical bar in Figure 7.6). In contrast, at the regional scale, species complement one another from the diverse localities of the region. As a result, additional species involve a steady increase of the regional functioning (Figure 7.6).

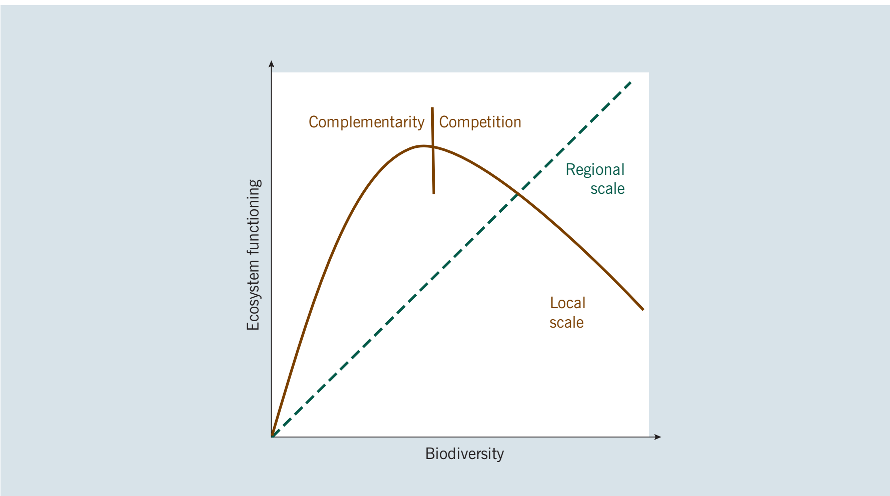


Figure 7.6: *B-EF relationships at the local and regional spatial scales. Local scale patterns cannot be directly transferred to regional scale because B-EF relationship may differ at both scales. According to these relationships, local species extinction may lead to a local improvement on ecosystem functioning up to a certain threshold (vertical line) whereas regional species extinction is immediately detrimental to ecosystem functioning*

Source: Redrawn from Bond and Chase (2002).

7.2. What do we know from stream ecology?

7.2.1. The specificity of fluvial habitats

When walking up a stream, we can easily catch a fundamental property of running waters, i.e. movement. As a fundamental part of the water cycle, the surplus precipitations that fall upon the continent (runoff) flow into the ocean allowing a permanent water turnover on Earth. When further scrutinizing the streambed at low water level, we find an amazing mosaic of different habitats (like sand, boulders, twigs, fallen leaves, algae) (Figure 7.7). The amount and the seasonality of flow induces a diversity of flow forces on the streambed, which in turn, affects the type and size of substrata (sand, cobbles, boulders) and the distribution of resources (twigs, fallen leaves, algae, and others).

In addition, stream flow implies that nutrients, dead organic matter (Figure 7.8), sediment and propagules are transferred from up to downstream as well as to the side arms in the floodplain, which makes streams and rivers open flowing systems in comparison to lakes or reservoirs. In the upper course of rivers, a large amount of the energy supply comes from the processing of dead organic matter, which originates from outside the stream channel. Part of this dead matter is processed locally by various species and another part is carried away downstream by flow. In contrast, lower courses are less affected by riparian shading and depend more on in-channel primary productivity and the organic matter coming from upstream involving different species to occur. In addition, flow and other environmental characteristics select species according to their

Figure 7.7: *Rivers flowing gently and showing the mosaic of habitats on the bottom substrate*

Figure 7.8:
A leaf pack in a French temperate river. Rivers transport leaves until they deposit and are processed by the stream biota (especially fungi and invertebrates). Leaf litter decomposition is one of the most studied river ecosystem functions

traits (see Box 7.1). Thus, flow is in rivers the most important driving force that affects its biodiversity and functioning.

7.2.2. Functional guilds and functional redundancy

Functional groups or guilds are groups of organisms that are believed to play the same role in ecosystems. For example, stream invertebrates can be divided, among others, into predators or shredders (animals eating large portions of dead organic matter). Cummins (1973) proposed to establish feeding guilds, known as functional feeding groups (FFG), mainly based on the mechanisms of feeding used by stream invertebrates and secondarily on the main type of food source (Figure 7.9). He, thus, implicitly recognized that knowledge on the functional role of species in streams should improve our understanding of the aquatic ecosystem functioning. Cummins (1973) especially noted that a majority of stream invertebrate consumers exhibited overlaps in their diets (i.e. they showed some functional redundancy see Box 7.3). For example, a detritivore may eat dead leaves but may also absorb small crustaceans or small insect larvae in some period. Similarly, an herbivore that usually grazes algae can potentially get dead organic matter in its food as well. In fact, stream insects cannot con-

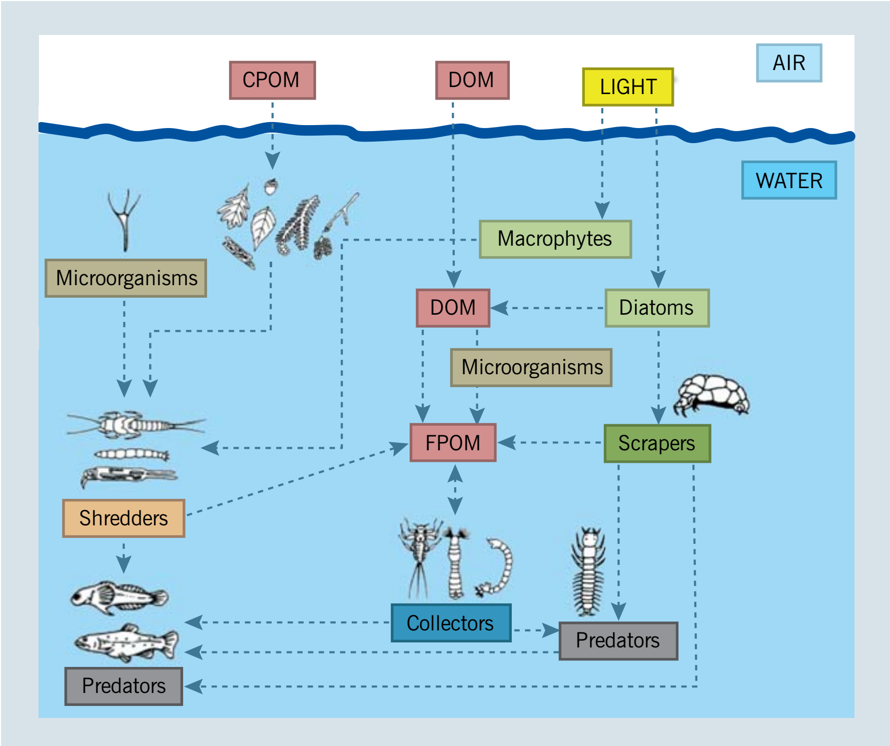


Figure 7.9: *The various functional feeding groups (FFG) that interact to process the matter and energy in streams and rivers. As indicated in this diagram, any FFG covers various life forms. All these life forms depend on each other and on energy inputs (CPOM stands for Coarse Particulate Organic Matter, FPOM for Fine Particulate Organic Matter, DOM for Dissolved Organic Matter)*

Source: Redrawn from Cummins (1973).

trol the food they receive, and thus, must be rather polyphagous and somehow opportunistic to get their food. This contrasts a lot with so many terrestrial insects, which are restricted to eating a single or a few plant species, as is the case of many caterpillars, for example. Moreover, diets may vary according to larval stages. For example, young *Hydropsyche* larvae (see Figure 7.12) mainly feed on algae and dead organic matter whereas older larvae may still feed on algae and also on other small invertebrates.

However, organisms within a given trophic guild are not necessarily redundant, as they may differ in their ecological requirements. For example, manipulative experiments with eight species of burrowing filter-feeding bivalves (freshwater mussels) showed that in summer one mussel species *(Actinonaias ligamentina)* reached greater biomass and had a higher excretion rate than other mussel species. In these conditions *A. ligamentina* benefitted benthic algae, which took advantage of the nitrogen excreted by the mussel (Figure 7.9). These differences between mussel species disappeared in periods of lower temperatures (Vaughn et al. 2007).

One species within a given FFG may thus not totally replace another species of the same FFG and the real degree of redundancy among species is far from being known. For example, it has been shown experimentally that three stonefly species belonging to the same FFG, such as shredders for example, had a different impact on leaf mass loss (namely, leaf litter decomposition; Figure 7.10). In this case, the same number of individuals was introduced in microcosms with each species alone, and grouped by two or three species together. In species alone situations, *Taeniopteryx nebulosa* had the highest impact on leaf litter decomposition, followed by *Nemoura avicularis*, and *Protonemura meyeri* (Figure 7.10 *left*). By contrast, decomposition increased with the number of species involved (Figure 7.10 *right*). Since the same number of individuals was used, these experiments suggest facilitation among species for processing leaf litter. As a result, at least experimentally, within-guild leaf litter decomposition rates can be significantly affected by the number of species belonging to the same FFG (Jonsson and Malmqvist 2000).

7.2.3. Context-dependency of the relationships between biodiversity and ecosystem functioning

The relationship between biodiversity and ecosystem functioning can be context dependent, as it may vary across the space. In other terms, some areas of the distribution range of a given species can be more suitable than others for its survival, growth and reproduction, and thus, its ecological effects there can be

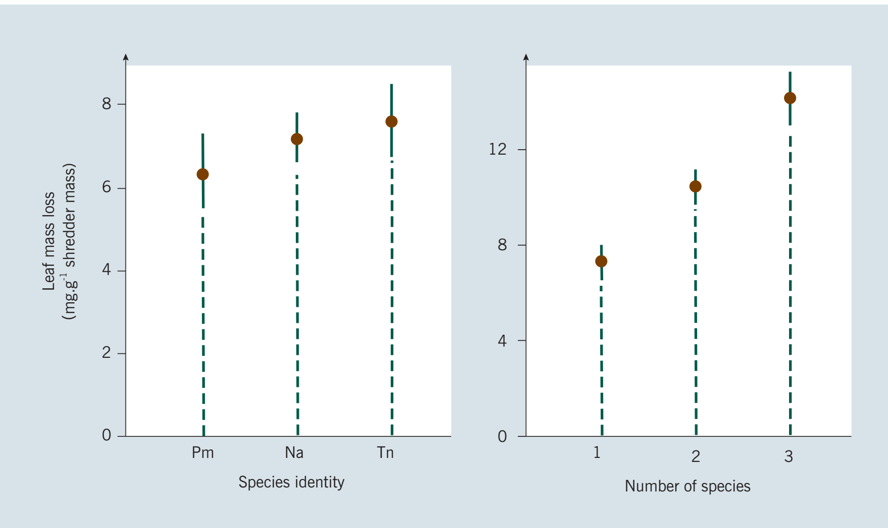


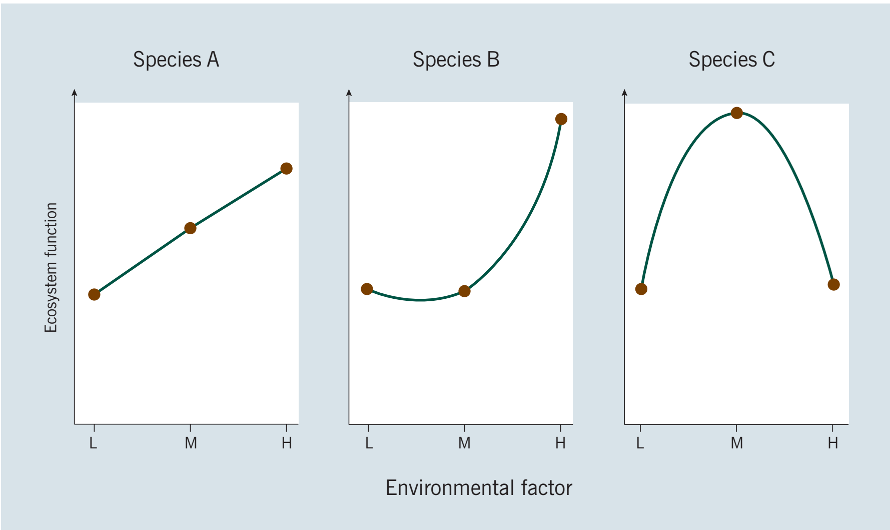
Figure 7.10: *Variation in leaf mass loss (mean ±1 SE) due to the identity of stonefly species feeding on dead organic matter* (left), *and with different species numbers* (right). *On the left, four individuals of each species alone were placed in microcosms, and decomposition increased from* Protonemura meyeri *(Pm) to* Nemoura avicularis *(Na) and* Taeniopteryx nebulosa *(Tn). On the right, grouping species together (by 2 or 3) increased the decomposition efficiency in comparison to species alone*

Source: Redrawn from Jonsson and Malmqvist (2000).

more important. For example, it has been shown experimentally that the effect of grazing insects on algal biomass changes with local current velocity (Figure 7.9). *Glossosoma verdona* (caddisfly) and *Baetis bicaudatus* (mayfly) ate less algae at slow and medium currents in comparison to fast current. In contrast, *Drunella grandis* (mayfly) had a strong effect on algal growth irrespective of current velocity. At fast current the three species had an equivalent impact on algae whereas at slow current, *D. grandis* had significantly greater impact than *B. bicaudatus*, and this one greater than *G. verdona* (Poff et al. 2003). Therefore, in some environmental conditions (in this case, low current velocity), different species may have a similar effect on ecosystem function (in this case, consumption of algae), and thus, appear as redundant, whereas the same species may differ in their effects when environmental conditions change (Figure 7.11). In other words, species with a strong effect on ecosystem processes in certain environmental conditions can become weak contributors under other environmental conditions, and this property may apply at different spatial and temporal scales.

Context dependency also relates to the degree and type of disturbances (Box 7.1). Biotic and abiotic disturbances (like grazing, predation, floods) causing mortality to organisms are key ecological factors that moderate the relationships between biodiversity and ecosystem functioning. As an example, net-spinning caddisfly larvae of the family Hydropsychidae are common in streams and feed on the dead particulate organic matter (POM) and small living organisms that drift in the water column with current. To catch their

Figure 7.11: *Hypothetical example showing how the performance of species belonging to the same guild and thus having a similar effect on an ecosystem function, may change with increasing value of a given environmental factor (L = Low, M = Medium, H = High)*



Source: Redrawn from Wellnitz and Poff (2001).

food they build nets on the bottom substrate, which enables them to capture the particles that drift in the current (Figure 7.12). In case of high densities of a more competitive hydropsychid, the nets of larvae can create flow shading and modify the hydrodynamic conditions in the immediate surroundings, which may prevent other filter-feeding species getting POM. It has been shown in manipulative experiments that disturbance could moderate such effect. The experiments consisted of creating disturbance artificially by randomly removing larvae of three species and their nets (thus imitating flood effects). By reducing the flow shading effect of the competitively superior hydropsychid, such artificial removal allowed a higher taxonomic evenness. In other terms, the other species could settle more easily. The resulting more diverse assemblage of filter feeders captured a greater fraction of POM. In contrast, in the absence of such artificial disturbance, increasing species richness led to dominance of the com-

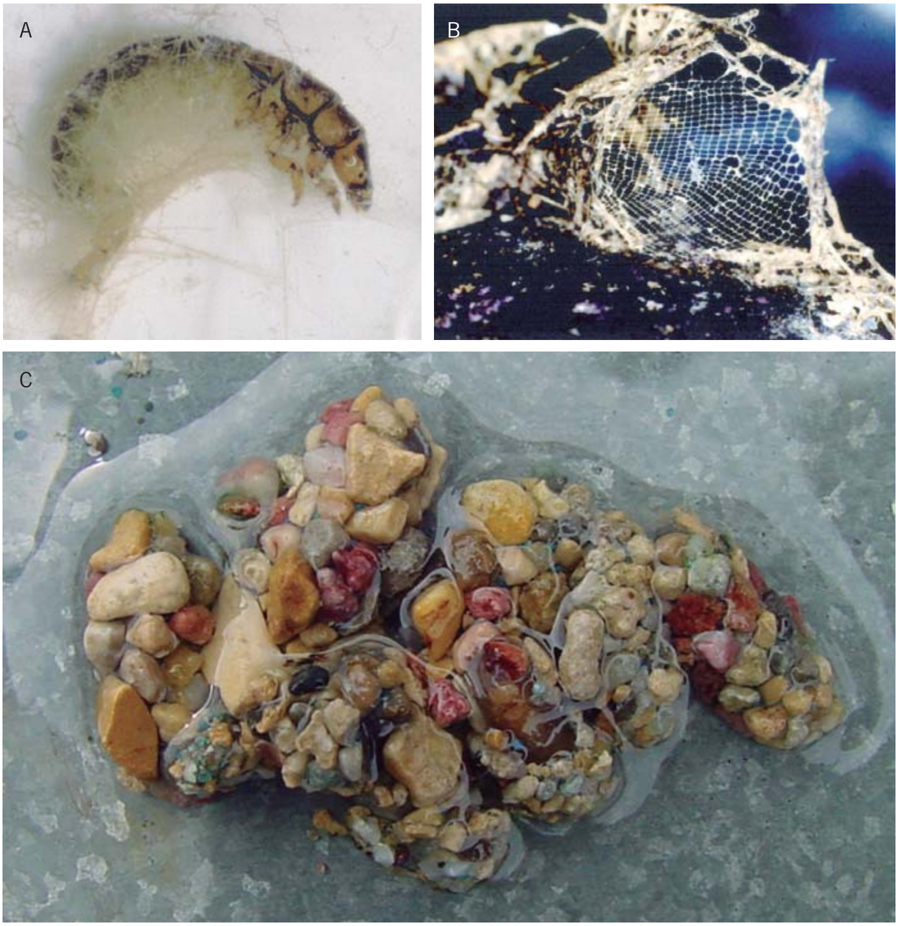


Figure 7.12:
A) A net-spinning caddisfly larvae (Hydropsyche exocellata) *surrounded by its net. B) The net itself constructed to capture particles drifting in the water column. C) A group of nymph cases with painted grain*

petitively superior hydropsychid and the amount of POM captured in water did not change when adding species.

Disturbance can also alter the indirect effect of net-spinning caddisfly larvae on other ecological processes such as algal productivity. For example, stream algal productivity partly relies on the amount of nutrients excreted by stream organisms since they use these nutrients as fertilizers. In the above manipulative experiments, it has been shown that in the absence of disturbance, namely when the competitively superior hydropsychid dominates the assemblage, the algal productivity declines. This apparently occurs because the prominent hydropsychid has particularly low rates of nutrient excretion.

Biotic and abiotic disturbance (Box 7.1) may influence ecosystem functioning in combination. For example, in the South Fork of Eel River (California) both floods and stocked fish can affect the abundance of insect larvae, and indirectly ecosystem functioning measured as algal productivity. In rainy years, floods slough insect larvae, and fish reduce the remaining insects, thus promoting algal growth. In contrast, during dry years, the insects are dominated by large armoured caddisfly grazers less vulnerable to fish predation, and algal biomass remains low (Power et al. 2008).

Disturbance can thus moderate the B-EF relationships by two mechanisms. One mechanism consists of a reduction of the effect of species with a disproportionate effect on ecological processes, like keystone species, ecosystem engineers, or species with biologically unique traits. The other mechanism relies on that preventing species dominance may result in increasing spatial heterogeneity, species richness and the rate of a given ecological process. In that case, to co-exist, species of the same guild have to differ by some amount in their biological traits so that they can feed in a complementary way on resources.

7.2.4. The importance of species dominance and identity

Most experimental studies addressing the B-EF relationship focus on species richness without considering the relative abundance of species within assemblages (evenness). In other terms, controlled experiments generally ignore that real local communities are usually dominated by few abundant species, which drive ecosystem processes and which coexist with many more rare species. However, besides species richness decline, human disturbances may produce changes in the relative abundance of species, which can greatly affect ecosystem functioning without noticeable change in species richness. For example, nutrient enrichment increases the abundance of a few species, which results in an increase of the ecosystem production. In contrast, siltation in streams fills in the interstices of the bottom substrata, which produces a decrease in the abundance of primary producers and therefore a decline of ecosystem production. In these two cases, measuring ecosystem functioning only through species richness would be misleading.

A common emblematic example of species controlling stream ecosystem functioning concerns salmon. A run of 20 million fish getting to spawning areas can move over 50,000 t of biomass into freshwater and adjacent terrestrial ecosystems. Salmon carcasses provide nutrients, which positively impact young salmon as well as a range of vertebrates and invertebrates that consume salmon resulting in high biodiversity. Overharvesting these migratory fish, thus, greatly disturbs the transport of materials over long distance and the chain bringing marine-derived nutrients to freshwater and terrestrial environments.

Biodiversity loss may affect ecosystem functioning. However, this effect depends on the degree and type of disturbances, the presence of dominant species and the order in which species are lost

Similarly, the relative abundance of shredders (as defined in Fig. 9) in assemblages may strongly influence the B-EF relationship in low-order streams where accumulation of leaf litter can be very important. It has been shown that for a given species richness, leaf litter decomposition was greater in communities with higher species dominance than in those with more even distribution of species. For instance, the crustacean *Gammarus fossarum* dominated the shredder community in a given stream throughout the year, and had a major impact on leaf litter decomposition even at low shredder diversity. In contrast, in other streams, breakdown rates peaked seasonally when two Trichopteran species dominated the community (*Sericostoma personatum* or *Chaetopteryx villosa*; Dangles and Malmqvist 2004). This example shows that species identity is a fundamental component of biodiversity with varying impact on ecosystem functioning. Hence, the biological traits of individual species strongly influence their abundance in communities and subsequently their roles in ecosystem functioning. For example, species such as *G. fossarum* showing strong specific interactions, high densities, present all year round in the stream and with a high mobility, namely able to drift and migrate upstream extensively can be expected to have strong effects on communities and ecosystem functioning. In contrast, the two above Trichopteran species demonstrate pronounced seasonal patterns in their biomass and resulting effect on decomposition, which shows in that case that the diversity effect on ecosystem functioning does not remain constant over time. As a result, deep knowledge of species identity and life cycles is mandatory for assessing further B-EF relationships and taking appropriate management measures.

7.2.5. The existence of positive interactions among species

As seen above, according to the competitive exclusion principle, two species relying on the same resources cannot coexist in a stable environment (see Figure 7.3). If one of the species has a slight advantage over the other then it will dominate, leading to either the extinction of the competitors or an evolutionary shift of their functional niche (niche differentiation). Such evolutionary shift may involve species feeding in a complementary way.

Besides the biodiversity and ecosystem functioning effects associated with such trophic niche differentiation and complementary use of resources, we should not forget positive interactions among species, which frequently occur in streams. Such positive interactions include aquatic fungi that condition leaf litter thus enhancing the palatability of leaves for shredders and initiating the detrital food chain. In addition, organism activities (e.g. for searching food, spawning, case building) contribute to sand and gravel transport or aggregation, thus, modifying both solid transport and biogeochemical processes on the streambed (two major stream functions), and the potential settlement of other species in the assemblage.

For example, net-spinning caddisfly larvae build their nymph case to metamorphose into adults (Figure 7.12). The grains that constitute the case are cemented together with silk similar to that used by spiders, and the case itself cemented onto the bottom substrate. At high densities, several cases can join together (Figure 7.12), and the ensemble can cause changes in the near-bottom flow forces. These caddisfly larvae can increase 9-fold the force necessary to mobilize gravel, thus stabilising the substrate and favouring the establishment of a diverse aquatic fauna and flora. However, here again appears complexity, since the locomotion activities of other species may be antagonistic. For example, gudgeon *(Gobio gobio)* and barbel *(Barbus barbus),* two species with different habitat preferences (near-bank gravel beds for gudgeon and coarse bed below riffles for barbel, a habitat similar to that of *Hydropsyche),* can reduce the flow forces necessary to mobilize gravels in different areas of a stream reach. We are yet far from a complete knowledge of the effect of species removal in such a complex context (Statzner et al. 2003).

The presence of a species may not only change the habitat for other species but may also affect the resource due to varying feeding efficiency. For example, it has been shown experimentally that the action of two detritivores on leaf decomposition could complement each other only if they were introduced in a well-defined sequence (Figure 7.13). The order in which the species colonize a given habitat is thus of critical importance for the functioning of the ecosystem.

7.2.6. The order in which species are lost do matter

While the sequence of colonisation of a stream reach by species affects ecosystem functioning, the sequential loss of taxonomically distinct invertebrate species

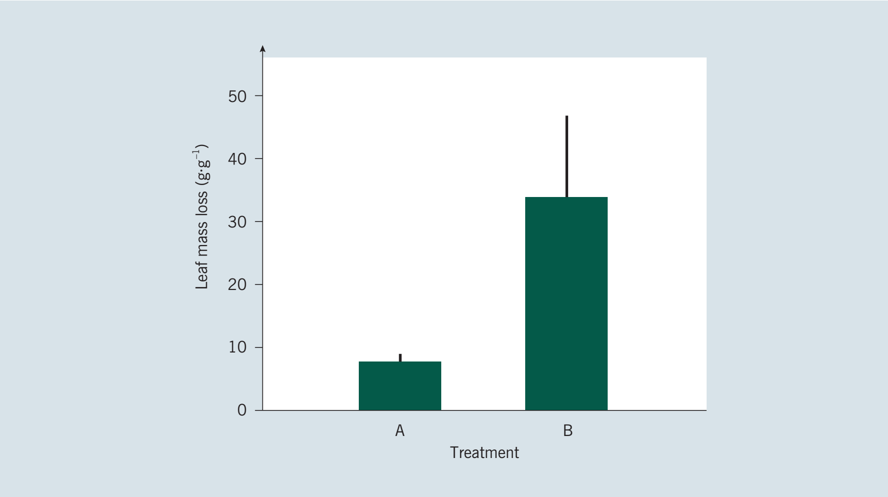


Figure 7.13: *Detritivore activity (leaf mass loss) of two shredding stoneflies introduced in sequence with (A)* Protonemura meyeri *following* Taeniopteryx nebulosa *and (B) the reverse option*

Source: Redrawn from Jonsson and Malmqvist (2003).

may also greatly affect ecosystem processes. Among the traits of species that do matter to changes in ecosystem functioning, body size has strong implications for organism metabolism. Small-sized organisms tend to have higher metabolic rates per mass unit than large-sized organisms. This can be measured by the *production-to-biomass ratio* (P/B), which represents the proportion of biomass produced by the individual of a species per time unit. Annual P/B ratio is higher for small-sized than for large-size organisms. In a modelling experiment that simulated species removal, it has been shown that the disappearance of all species of a given size class in sequences from large to small body sizes during repeated extinction events involved an increase of annual P/B (toward a value 5 times that of the initial entire assemblage; Figure 7.14). In contrast, if at each extinction event, the species are lost at random regardless of their body size, annual P/B remains relatively stable (Figure 7.14).

This simple example illustrates the outstanding importance of size in any consideration of species loss and function. Patterns similar to those simulated may occur when a stream receives pollution. In this case, diverse assemblages that include large invertebrates yielding high biomass and low production are replaced by species-poor assemblages dominated by small tolerant species with a low biomass and a high production. The order of extinction can easily be assessed for various types of anthropogenic disturbance. For instance, acidification affects mainly organisms sensitive to lack of calcium (crustaceans and molluscs), whereas organic pollution affects those sensitive to oxygen depletion. Therefore, species traits do matter for assessing the decline in ecosystem functioning.

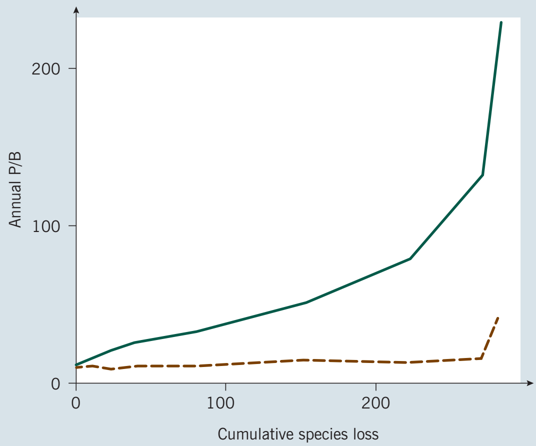


Figure 7.14: *Simulation of the effect of random species extinction independent of body size (dashed line) and of removing species sequentially from large to small body sizes (plain line) on annual production-to-biomass ratio (P/B) in hypothetical stream invertebrate assemblages. If large and intermediate size classes disappear, the annual P/B of the assemblage rapidly increases (plain line). If species go extinct at random, the annual P/B remains relatively stable with increasing species extinction, and increases only when a single species remains*

Source: Redrawn from Statzner and Moss (2004).

A further complication to bear in mind is that large animals, those likely to go extinct first, may belong to different trophic levels, such as detritivores and predators. The manipulation of large predatory invertebrates in experimental stream channel shows that their absence can promote grazers and reduce biomass of benthic algae, and even reduce sediment accumulation. The experimental exclusion of large detritivores in the same experimental channel affected both the magnitude and the rate of litter decomposition. Small-size detritivores are unable to compensate the lack of large detritivores, thus leading to a decline in leaf decomposition rate (Lecerf and Richardson 2011). As a result, large stream invertebrates may affect multiple ecosystem properties. As they will generally disappear first, their loss will critically affect ecosystem structure and functioning.

The consequences of species extinctions on ecosystem functioning thus depend on the species and its interaction with others in the food web. They may induce an increase of some species when their competitors and/or predators decline. The effects of biodiversity on ecosystem functioning may also depend on whether biodiversity loss occurs at a single trophic level, or at multiple trophic levels. From several studies covering various types of stream ecosystems, it has been shown that species richness had a weaker effect on ecosystem functions than assemblage composition of overall species, which indicates again the importance of species identity, species traits and functional diversity in comparison to taxonomic diversity. In addition, this meta-analysis showed that the species composition effect was found to be more pronounced on ecosystem function at lower trophic levels in comparison to species richness, whereas both the richness and composition of predators affected ecosystem functions equally (Lecerf and Richardson 2010). All these elements acting at different biological scales show how difficult it may be to accurately predict the effect of biodiversity loss on ecosystem functioning.

7.2.7. What about species gain?

Many human pressures involve species loss but also species gain, which derives from the establishment of non-indigenous species (deliberate or accidental introduction of organisms to an ecosystem; see chapter 8). Such biotic exchanges appear as one of the five most important determinants of changes in overall biodiversity together with changes in land use, atmospheric CO_2 concentration, nitrogen deposition and acid rain, and climate. In general, invasive species have traits (temperature tolerance, body size) that favour their establishment and population growth and may lead to the replacement of native by invasive species. However, the functional consequences of invasive species remain to be documented. For example, freshwater gammarids that are commonly considered as shredders (see above) and suggested to have a strong impact on leaf litter decomposition may exploit a wide food range. Now, the originally Ponto-Caspian gammarid *Dikerogammarus villosus* has invaded many European freshwaters where it is progressively eliminating native gammarids from European freshwaters through predation. In experimental flumes, *D. villosus* was able to withstand stronger currents than the native *Gammarus pulex*. Under high velocities, *G. pulex* tended to concentrate in flow refuges, thus being easy prey for *D. villosus* and resulting in increased mortality of *G. pulex*. However, leaf litter decomposition only moderately decreased in the presence of *D. villosus* (Felten et al. 2008) showing that the invasive species had a moderate effect on the ecosystem function. In contrast, due to their high densities, the signal crayfish *(Pacifastacus leniusclus)* has been shown to dramatically alter sediment transport thus deeply impacting ecosystem functioning (Harvey et al. 2011).

7.3. Take-home message

High species richness in streams results from an array of processes including the ability of species to cope with environmental conditions, their dispersal ability, and subtle interactions that allow them to coexist locally by partitioning the resources. Changes in species richness affect ecosystem functioning, but the species identity may matter much more than species richness per se. Looking at species richness alone may be thus misleading for addressing the effect of biodiversity loss on ecosystem functioning since changes in the abundance of some species might impact ecosystem function even in the absence of local extinctions. As an additional complexity, few scientific studies have clearly shown how the functional performance of species varies in different environmental conditions.

Rivers hold a huge biodiversity despite covering a little percentage of the Earth area. Conservation strategies should be prioritized in habitats having key species for ecosystem functioning

The role of non-trophic interactions among stream species also appears insufficiently appreciated. For instance, ecosystem engineers include beavers that build dams across rivers, thus strongly affecting their functioning. In fact, most stream species either consolidate or disturb the bed sediment, which has consequences not only on the bottom substrate mosaic, but also on resource fluxes and the establishment of other stream organisms. However, we lack evidence about the ecological consequences of removing engineer species, especially because some of them may involve bioturbation whereas in the same area other may consolidate stream. We currently do not know how the resulting antagonistic effect of both types of engineers may affect an ecosystem function such as bed sediment transport. To investigate the impact of human disturbances on ecosystem functioning, we need to establish scenarios of extinctions that are characteristic for a given type of disturbance and to consider the non-random sequential loss of species, which depends on the traits of species, among which body size is determinant.

Taking into account that predicting changes of ecological functioning from changes of biodiversity remains a complex task at regional scale (the scale at which environmental policies operate) and since most of the B-EF responses were assessed from local scale experiments, we should keep the B-EF hypothesis as a working hypothesis. B-EF tests suggest measuring biodiversity by taking into account the identity of species (their traits, their life cycles) rather than species richness alone. Once a few of such species have been recognized as keys for some ecosystem functioning in a given ecosystem then conservation measures should concentrate on preserving their environment. Preserving the environment of a key species means preserving the biotic and abiotic filters, which induce the preservation of other species and having an appropriate ecosystem functioning.

Currently, a straight match between biodiversity and ecosystem functioning in streams is thus far from obvious and urges B-EF scientists to develop new research combining field studies and laboratory experiments at different scales. It should also impulse managers to implement present scientific knowledge in conservation, management and restoration. A key implication of the B-EF hypothesis is, however, that the final target relies on receiving a service from ecosystems. Therefore, the B-EF hypothesis assumes that biodiversity should be preserved because it ensures a service rather than for its own intrinsic value. We should not forget that every form of life is unique, deserving attention regardless of the ecosystem service it provides to human society. Biodiversity is above all part of our natural and cultural heritage.

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Chapter 8

The Problem of Invasive Species in River Ecosystems

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Invasive nonnative species are a major problem in river ecosystems, and have large ecological and economic costs. Few ecosystems can resist invasions. The species that tend to invade most readily are those that humans introduce the most, and the ecosystems they invade are those with the most human activity. Most invasions are irreversible, and control is expensive, so efforts should be focused on prevention of future invasions.

8.1. The problem of biological invasions in rivers

The number of new species invading ecosystems has exploded in the last 50 years, primarily because of the increase in human transportation and shipping. These forces have broken down natural barriers that previously prevented organisms from moving around the globe. Although humans have brought plants and animals of interest to new regions for thousands of years, the scale and speed of these introductions has recently increased exponentially, similar to the pace of human population growth and resource use.

Biological invasions have been especially rapid in freshwater ecosystems, where nonnative species now make up a substantial proportion of the fauna in many regions (Moyle and Marchetti 2006). For example, across large regions like the Pacific Northwest of North America, southwestern Europe, central Eurasia, South Africa, and southern Australia, nonnative freshwater fish make up more than a quarter of all fish species in river basins and in some cases up to 95% (Leprieur et al. 2008). In contrast, in most continental regions of similar size the percentage of nonnative plants is < 25% (Vitousek et al. 1996).

Why are invasions so prevalent in freshwaters, including rivers and streams? First, most aquatic habitats have been highly modified by human actions, reducing native species and creating conditions suitable for tolerant nonnative fishes (Rahel 2003). Second, introductions of fishes and other freshwater organisms have been common and frequent, both intentionally for food or sport and inadvertently by creating canals or other connections between waterways. Larvae of most fish and invertebrates are tiny, and so are easily transported without notice when ships release ballast water, or when fish are stocked from hatcheries. In addition, anglers are now illegally introducing many species that they value highly for sport fishing.

But how important can these freshwater invasions be? After all, many people enjoy catching and eating fish, whether they are native or nonnative. Unfortunately, invasions in freshwater ecosystems can cause extensive problems for regional economies, human health, and the integrity of ecosystems. Many of the largest losses are caused by invertebrates that live in freshwaters for at least part of their life cycle. Asian tiger mosquitos *(Aedes albopictus),* which have invaded North America and Europe, are vectors for the viruses that cause dengue and yellow fever, among the most important human diseases in the tropics. The zebra mussel *(Dreissena polymorpha),* originally native to the Black, Caspian, and Azov seas, invaded North America via ballast water releases and Europe by a combination of pathways, including canals and shipping. These tiny mussels clog water intakes, and require hundreds of millions of US dollars to control (Strayer 2009). In addition, their filtering is altering river and lake ecosystems, by reducing phytoplankton and increasing macrophyte biomass, thereby altering entire food chains that support important commercial and sport fisheries. The introduction of Nile perch *(Lates niloticus)* in Lake Victoria is thought to have caused the greatest modern human-caused mass extinction of vertebrates by extirpating dozens of endemic cichlid fishes through predation. It is clear that invasions in freshwaters are major problems.

Ecologists now agree that nonnative species invasions are among the leading causes of biodiversity loss worldwide, falling not far behind direct habitat destruction (Vitousek et al. 1996). The world's freshwaters make up a tiny fraction of the water on earth, but support a large proportion of aquatic biodiversity, primarily

Figure 8.1: *Aquatic invaders can cause extensive damage to ecosystems, human health, and regional economies. Clockwise from upper left: Asian tiger mosquito, zebra mussels, rainbow trout, Nile perch*

because their relative isolation allowed many more species to evolve (Dudgeon et al. 2006). Moreover, the loss of biodiversity is higher for organisms that inhabit freshwater ecosystems compared to terrestrial ones. For example, between a third and three-quarters of all fishes, crayfishes, and freshwater mussels in the U.S. are imperiled or extinct. Together, these two facts suggest that nonnative species invasions are a greater problem in freshwaters than terrestrial or marine ecosystems.

In this chapter we review this important issue in river conservation, focusing on the principles governing invasions in flowing-water ecosystems and how human-caused stressors change the causes and effects of invasions. We show that invaders can have important effects on many levels, from local extinction of native species to changes in ecosystem services. We compare the patterns of invasion in two highly-invaded regions, namely the Iberian Peninsula and the Colorado River Basin, and consider lessons learned. Finally, we summarize the implications of these lessons for river conservation, and discuss priorities for management. We focus on inland fishes but also provide examples from of other groups of freshwater organisms.

8.2. Principles for invasions in river ecosystems

Ecologists have worked for many decades to answer two main questions about biological invasions of most interest to managers: 1) Which ecosystems are most likely to be invaded?; and 2) Which species are likely to invade next? The upshot of all this research is that invasions are not easy to predict, there are typically no simple answers, and some answers are paradoxical. Nevertheless, some general principles have emerged for aquatic ecosystems, including rivers, lakes, and reservoirs.

First, analyses for several large areas in North America show that virtually all aquatic ecosystems, even those with many native species already, can be invaded (Moyle and Marchetti 2006). There is little evidence that those with more native species are somehow “saturated” and can resist invasions by nonnatives (Gido and Brown 1999). This is perhaps not surprising, because freshwaters worldwide support about 126,000 (9.5%) of the more than 1.3 million animal species described, including about 12,500 fish species. Many new species are available to enter every ecosystem.

Although all ecosystems are invasible, several factors tend to foster invasions in some more than others. Paradoxically, these have more to do with human factors than characteristics of the ecosystems or the species themselves. For example, lakes and reservoirs have been a main target of fish introductions, because anglers prefer to fish there, people enjoy them for recreation, and because reservoirs are wrongly seen as creating “empty ecological niches” that should be stocked with fish or invertebrates. However, reservoirs are nearly always fed or drained by rivers, as are many lakes, creating a perfect source for nonnative species invasions into flowing waters. Many fish invasions into reservoirs or headwater lakes have spread throughout entire river basins by connecting channels, especially in the downstream direction. And, because reservoirs are often the target for stocking, rivers and streams that are impounded by dams often have higher richness and abundance of invasive species (Marchetti et al. 2004). In contrast, some headwater streams have fewer nonnative species, either because they have received fewer introductions, or because natural or artificial barriers have prevented invaders from moving upstream into them (Fausch et al. 2009).

Ecologists have spent many years attempting to predict which fish and invertebrates will invade different aquatic ecosystems, the second main question of interest. Many biological traits of nonnative species, such as temperature tolerance or body size, can be important determinants of their invasion success in particular water bodies. However, the best predictors for invasions are usually simply those species that are of interest to humans, and either are readily available or occur nearby. For example, rainbow trout *(Oncorhynchus mykiss)* have been introduced to nearly 100 countries worldwide for angling and are cultured in many for food. As a result, this species is much more likely to be introduced and invade in coldwater streams than many other small coldwater fishes which are restricted to certain regions and are never cultured or angled. In contrast, zebra mussels, New Zealand mudsnails *(Potamopyrgus antipodarum),* and the invasive algae *Didymosphenia geminata*, are more likely to be transported by humans on boats or gear to nearby waters than to those farther away, although long-distance transport is certainly possible.

One of the best predictors of which species will invade is simply those that are introduced most often (Simberloff 2009). Ecologists use the term "propagule pressure", a combination of the number of organisms introduced and the number of times introductions are made. Propagule pressure is typically very high for introductions in aquatic systems compared to many terrestrial ones, for several reasons. First, large numbers of tiny fish and invertebrate larvae can be carried inadvertently when water is transferred among natural water bodies. Second, aquatic organisms like fish have many eggs and tend to be easy to raise in aquaculture, compared to birds or mammals, so many thousands can be produced for stocking. A third is that because freshwater ecosystems provide so many important ecological services to humans (recreation, transportation, irrigation), both water and fish are often transferred among them, increasing the frequency that nonnative organisms are introduced.

Invading species not only drive native species locally extinct, but can also reduce the flow of nutrients and food resources in river food webs, and even into the riparian zone

Human interest and activity often drive invasions (Marchetti et al. 2004). In general, humans introduce a very small percentage of the 12,500 freshwater fish species, focusing on those that are preferred or have been successful in the past, such as brown trout *(Salmo trutta),* common carp *(Cyprinus carpio),* and mosquitofishes *(Gambusia affinis and G. holbrooki).* These species are known to survive and reproduce in many waters, so they make up a small set of cosmopolitan species that are of interest to humans and become frequent invaders. In addition, because many introductions of fish to rivers are intentional, and nowadays often illegal, people that introduce them know well where certain species will be able to survive and reproduce. In particular, illegal stocking by anglers has recently become the most potent source of fish invasions (see Box 8.1). As a result, on a global scale human activity is a better explanation for where fish invasions occur than are either the number of native species that could resist invasions, or the characteristics of the environment that might resist or foster them (Leprieur et al. 2008).

Figure 8.2: *Illegal stocking by anglers of predators like the European catfish* (Silurus glanis) *is now a major source of fish invasions*

A final principal common to all invasions is a key paradox. Although few of the species that are transported to new locations become established and cause damage, those few that do can be very costly. Most species that arrive in new regions are not introduced to the wild, most species that are introduced do not establish, and most that establish do not become pests. However, costs for damage and control of invasive species were already hundreds of billions of dollars for a small subset of developed countries a decade ago (Simberloff et al. 2005), and can only rise. In addition, because many introductions to freshwaters are intentional (e.g. fish and crayfish for food or angling), propagule pressure is high, and humans are able to match habitats closely for well-known invaders, invasions in aquatic ecosystems are becoming a major force in global ecological change.

Biotic homogenization is a relatively new term used to describe the spread and dominance by the relatively small set of cosmopolitan species that humans introduce to other ecosystems (Olden 2006). Loss of rare species found only in specific locations, called endemic species, also contributes to this pattern of the increasing similarity of the earth's biota. However, most of this increasing similarity among freshwater fishes is driven by the invasion and spread of nonnative species, rather than the local extinction of rare endemics (Gido and Brown 1999; Rahel 2003). For example, the 48 states in the conterminous U.S. now share 15.4 more species on average that they did originally (a 7%

Box 8.1

Illegal fish stocking by anglers is the most important new source of intentional fish invasions

During 1850-1980, most of the purposeful introductions of fish to new waters throughout the world were conducted by government fisheries management agencies. However, given the burgeoning problems with invasive species, and better education and awareness, most fisheries management agencies have sharply curtailed introduction of nonnative species. An exception to this is the continued stocking of nonnative trout, especially rainbow trout, which in some regions are treated as a native or naturalized species.

However, a major new source of purposeful introductions is from illegal stocking by anglers. For example, unauthorized introductions in seven regions throughout the USA made up 90% of new fish introductions during 1981-1999, compared to only 15-43% during all previous periods. Similar trends are apparent in Europe and Australia (Johnson et al. 2009). Live fish wells in boats coupled with sophisticated knowledge and communication by anglers have made it easier than ever to move live game or bait fish to new bodies of water where anglers perceive that sport fisheries could be improved by introducing nonnative fishes.

Why do anglers stock illegally? One reason may be that anglers assume that stocking is not a problem, given that fisheries management agencies also have stocked nonnative fishes in the past, and sometimes do currently (Johnson et al. 2009). In addition, public education of the risks of stocking is often only rudimentary, and penalties are modest, averaging less than US$3,000 across 12 western states where problems are the greatest. Finally, agencies often fail to respond strongly to illegal stocking, in some cases even setting angling regulations to encourage sport fishing for the new species. This can also encourage angler groups that advocate for the new game fish, making future eradication politically impossible.

What problems does illegal stocking cause? Like other invasions by aquatic organisms, illegally stocked game or forage fish cause large losses to native fish assemblages, and to established sport fisheries based on managed or unmanaged nonnative fishes. When new nonnative species become established, they also provide sources for invasions elsewhere, and foster more illegal stocking. Costs for eradication or control of the nonnatives are huge. Eradication of nonnative northern pike *(Esox lucius)* twice from one California lake cost US$33 million. Lake trout *(Salvelinus namaycush)* illegally stocked by anglers invaded Yellowstone Lake in Yellowstone National Park, and the lost revenue from fisheries for native trout alone will approach US$1 billion over 30 years, not to mention the US$300,000 per year for lake trout eradication. Likewise, the invasions themselves can invalidate recovery programs for threatened native fish species that cost tens of millions of US dollars (Johnson et al. 2009).

What can be done to reduce illegal stocking? Fisheries management agencies will need to set responsible policies for their

Box 8.1 *(cont.)*: Illegal fish stocking by anglers is the most important new source of intentional fish invasions

own stocking of nonnative fishes, and communicate the reasons for these to the public. They will also need to set uniformly strict regulations against illegal stocking, and holding or transporting live fish, as well as impose large fines that reflect the huge economic costs that illegal stocking causes. For example, Canada imposes a maximum fine of US$100,000 for illegal stocking, which is still far short of the cost required for eradication in most waters (Johnson et al. 2009). For anglers who are unaware of the risks, articulate and balanced messages that are widely distributed can help curb illegal stocking (see: http://stopstocking.cowyafs.org/). However, for those prone to vandalize waters, severe sanctions, large rewards for witnesses, and showcasing convictions will be needed to reduce the impetus for these acts.

increase in similarity), owing to introductions for sport fishing or aquaculture (Rahel 2003). Common carp, goldfish *(Carassius auratus),* brown trout, and rainbow trout are the species most widely introduced in this region. Similarly, fish assemblages in the Iberian Peninsula are 17% more similar now than originally, owing primarily to introductions from France of top predator fishes, like European catfish (Clavero and Garcia-Berthou 2006). Overall, these invasions by cosmopolitan species support the contention that few aquatic systems are “saturated” with species, and invaders that are pre-adapted for either natural or altered conditions are likely to invade if transported and released in sufficient numbers.

8.3. Human stressors that can change the outcome of invasions

The principles described above help ecologists understand and predict the general patterns of invasions in rivers, but other anthropogenic stressors can change the outcome of species introductions in specific locations. Ecologists are well aware that natural environmental factors can affect whether nonnative species become established, and how strongly they affect other species. For example, invasions of nonnative trout in North America are more common in regions where the seasonal flooding regime matches that in their native rivers than in regions where they do not match (Fausch 2008). Here we focus instead on how anthropogenic stressors can potentially change the outcome of invasions. English ecologist Charles Elton, who pioneered the field of invasion biology, gave the first example of this when he reported that invasions are common in habitats that have been degraded by humans, although many ecologists forget he also pointed out that species can invade pristine habitats. Nevertheless, various ecologists have inferred from Elton’s idea that restoring habitat quality and natural processes may help reduce nonnative species abundance, or the risk of future invasions.

8.3.1. Altered flow regimes

Natural patterns of stream flow are expected to favor native species that are adapted to the natural "disturbance regime", whereas nonnative species may be favored when hydrologic conditions are altered (Poff et al. 1997). For example, nonnative fishes are reduced by natural flash floods in southwestern USA desert streams, apparently because they lack appropriate capacity for seeking refuges. Mosquitofish are an aggressive predator that can extirpate native topminnows, and were favored in desert streams where floods were damped by hydrologic alteration. Similarly, nonnative fish were reduced in years of higher summer flows in a central California river with Mediterranean climate. However, years with lower flows and warmer temperatures favored nonnatives like largemouth bass *(Micropterus salmoides)* and common carp that spawned during summer. Nonnative crayfish were also reduced by natural floods in an eastern California mountain watershed.

Altered flows can also shift the balance for native and invading riparian plants, although the effects differ between them. Nonnative tamarisk *(Tamarix ramosissima)* has invaded many rivers of the southwestern USA, whereas native cottonwood *(Populus deltoides)* has simultaneously declined. Cottonwood seeds require moist bare sand created by natural floods to germinate, and high water tables caused by the floods to survive as seedlings. A wide-ranging comparative study showed that cottonwood seedlings are very sensitive to altered flows, and declined to low levels even with modest flow alteration (Merritt and Poff 2010). In contrast, tamarisk can invade under altered flows, because its seedlings can survive under more variable flow conditions than cottonwood. However, tamarisk can also invade under natural flow regimes, wherein the two species may achieve about equal abundance. Elton suggested that this scenario may be common, where natives may find at least partial refuge under natural conditions.

Paradoxically, natural droughts may also provide some protection for native species in arid climates. A native galaxiid fish, which is similar to trout, persisted better in the naturally intermittent flow regimes of headwater streams in south-central Australia. Nonnative trout died out under these conditions, but persisted better in downstream reaches with more constant flow. However, in southcentral USA streams, nonnative crayfish persisted better than native crayfish under drought, which experiments showed was owing to their greater tolerance to drying. Such drying may become more frequent with climate change.

These examples demonstrate that natural flow regimes may favor native species, but the effects are often complex. For example, nonnative species may be able to invade even under natural flows, as for tamarisk, or nonnatives may be more tolerant of periodic disturbances like drought, as for certain crayfish. Likewise, the ability of nonnative species to displace native species via competition, predation, or disease can interact in complex ways with changes in flow or temperature (Wenger et al. 2011). Therefore, predicting whether restoring natural flow regimes can favor native species over invaders, or how stressors like climate change will affect invasions, will require careful consideration and testing of such mechanisms (Rahel and Olden 2008).

8.3.2. Habitat alteration

Ecologists have consistently found that fish invasions are higher in areas with more human habitat degradation, such as from urbanization, transportation, and mining. This pattern is apparent in regions ranging from California, the lower Colorado River Basin USA, Australia, and across the world (e.g. Marchetti et al. 2004; Leprieur et al. 2008). However, these authors suggest that the actual mechanisms causing invasions are increased releases from unwanted aquarium fishes, bait buckets, ballast water, and intentional introductions, which are a by-product of higher urban and suburban development. Instead of habitat alteration itself promoting invasions, it may be simply that more species are introduced actively or passively in more disturbed habitats where more humans live.

8.3.3. Climate change

Increased warming and variability of the climate is compounding other stressors like altered flow regimes and habitat degradation. Most studies of climate change have focused on the effects of increased temperature, especially for coldwater fishes, and large losses are projected for native trout and charr under typical warming scenarios. However, flow regimes are also predicted to change, often from snow to rain in mountain regions, and these may combine with temperature and species interactions to drive outcomes (Rahel and Olden 2008).

For example, cutthroat trout *(O. clarkii),* native to the western USA, are strongly depressed by nonnative brook trout *(Salvelinus fontinalis),* brown trout, and rainbow trout. However, these nonnative trout themselves are predicted to decline from increased water temperatures, and fall-spawning brook and brown trout are susceptible to increased winter floods that can wash away their spawning nests or newly-emerged fry (Fausch 2008). In contrast, rainbow trout and the native cutthroat trout spawn in early summer and so

Figure 8.3:
Habitat for trout in Rocky Mountain rivers of the western USA is predicted to decline by half by 2080 as the climate changes, but effects will be stronger for some nonnative trout than native species

are little affected by winter floods. A detailed analysis throughout the Rocky Mountains using the latest climate and flow predictions showed that trout habitat will decline by nearly half in 70 years (2080) from a combination of these effects (Wenger et al. 2011). Paradoxically, habitat for nonnative brook trout will decline more than for cutthroat trout (77% *vs.* 58%), owing to warmer, rainier winters, making the situation for the native trout a bit better than it would have been otherwise. Nonnative rainbow trout are predicted to decline the least of all the trout (35%), because negative effects of increased temperature are partly offset by positive effects of more favorable flow regimes. Therefore, climate change will likely result in complicated interactions among several factors, all of which must be considered simultaneously to make accurate predictions.

8.4. Invasions cause effects at multiple levels in ecosystems

In addition to predicting future invasions, ecologists have recently become more interested in determining the effects of invasions at multiple levels, from other species to whole ecosystems. Some invaders cause declines and even extinctions of other species through predation or competition. For example, mos-

Figure 8.4: *Nonnative brook trout* (upper left) *forage more on bottom-dwelling stream insects than native cutthroat trout* (lower left) *in streams of the western USA, ultimately reducing emerging adult insects that feed riparian spiders* (right), *as well as birds, bats, and lizards (see Box 8.2)*

quitofish and two trout are listed among the 100 world's worst invasive species, and have caused many local extinctions of other fish species.

However, invaders may also affect communities and ecosystems in other ways. For example, species are often finely tuned to each other via natural selection, as predators and prey, or competitors. When an invader takes over, it can change these selective forces, and so cause native species to evolve different characteristics in response. In addition, the invaders themselves change genetically as they integrate into the new ecosystem and leave their natural enemies behind.

Invaders like zebra mussels can also alter ecosystem services, such as the way that nutrients like nitrogen and phosphorus are moved from river sediments into the water column and made available for plant growth (Strayer 2009). One of the most interesting cases is of nonnative trout, which can reduce the abundance of bottom-dwelling stream insects by their own foraging or by altering the foraging of native trout. This causes a cascading set of changes in the stream food web, which ultimately reduces the abundance of adult insects that emerge from the streams and become prey for streamside predators like birds, bats, lizards, and spiders (see Box 8.2).

8.5. An intercontinental comparison: The Colorado River and Iberian Peninsula

Case studies can be useful for comparing sources and patterns of invasions, and conservation challenges. Here we contrast the Colorado River Basin (CR)

Box 8.2

Ecological surprises: Can nonnative fish in streams affect birds and spiders in the streamside forest?

Ecologists have long known that invertebrates that fall or blow into streams from streamside (riparian) forests or grasslands are valuable food for fish. However, they have only recently measured the emergence of adult aquatic insects like mayflies into the riparian zone, and discovered that many terrestrial animals there make a living on these insects that start life in the stream. For example, about half the food energy that fish in small streams need comes from insects that fall in from the land, and more than a quarter of the energy that riparian birds need can come from insects emerging from small streams. Emerging insects also provide much of the diet for riparian bats, lizards, and spiders, especially in early spring when most insects emerge from streams.

As it turns out, nonnative trout can strongly reduce this insect emergence. In turn, this can reduce spiders, and potentially other riparian animals like birds and bats, through a cascading series of changes in the stream-riparian food web. For example, adding nonnative rainbow trout to reaches with native Dolly Varden charr in a northern Japan stream caused the charr to switch their feeding to bottom-dwelling insect larvae. In turn, this reduced the abundance of the emerging adult insects by a third, which reduced the abundance of spiders in the riparian zone by two thirds (Baxter et al. 2004). Cutting off the emergence entirely using a mesh greenhouse reduced spiders by about 85%, so the effect of rainbow trout was indeed strong by comparison.

Nonnative brook trout in Rocky Mountain streams of the western USA have similar effects when they replace native cutthroat trout (rather than being added to them as in the Japan study). Brook trout forage more on the bottom-dwelling insects, and two studies showed that they reduced the biomass of emerging insects by between a third to a half compared to the native trout (e.g. Benjamin et al. 2011). This reduction in emergence was projected to reduce riparian spider abundance by 6-20%.

These studies are part of a growing body of evidence that nonnative species can have strong, unexpected ecological effects, such as on emerging insects that form strong linkages between streams and riparian zones. More importantly, these effects can cross habitat boundaries to create ecological surprises in distant locations. Nonnative trout can indeed affect riparian birds, as well as other riparian animals that people care about.

in the southwestern USA with the Iberian Peninsula (IP; Spain and Portugal) in southwest Europe. Both are generally arid regions, which have been subject to many fish introductions and invasions. Our main goal is to seek common patterns between the regions, and differences, and to highlight the importance of the topics discussed previously. We focus on fish invasions because data and research are more complete, compared to invasions of plants or invertebrates.

8.5.1. Basin characteristics and human stressors

The two regions are of roughly similar area, and both drain from high mountains to the sea (Table 8.1). Climate is generally dry, although parts of the Iberian Peninsula are wetter or more arid than the rest. The Colorado River Basin generally has high flows during summer from melting snow, whereas most Ibe-

Table 8.1: *Intercontinental comparison of fish invasions in the Colorado River Basin in the southwestern USA, and the Iberian Peninsula in southwestern Europe*

Characteristic	Colorado River Basin, USA	Iberian Peninsula
Area (km^2)	639,000	581,000
Maximum elevation (m)	3105	3479
Climate; flow regime	Continental; summer snowmelt	Primarily Mediterranean; mostly autumn-winter rain
Primary/secondary habitat alterations	Dams and flow regulation/ channelization	Dams and flow regulation/ water abstraction
Number of native freshwater fishes (endemics; % of total)	35 (24; 69%)	51[a] (41; 80%)
Imperiled[b] native species (% of total)	20 (57%)	49 (96%)
Number of established nonnative species (total; invasive)	72; 29	26; 12
Native region of most nonnative species (% of total)	Mississippi River Basin (67%)	Europe (42%)
Sources of most introductions	Stocking for fisheries management, bait minnow releases	Illegal stocking for angling, aquaculture
Prevention efforts	Eliminated stocking (except salmonids) Restricted use of bait minnows Boat inspections to prevent transporting invasive species	Black list of invasive species Restricted navigation in zebra mussel infested waters Outreach on the zebra mussel
Control and eradication efforts	Removal of nonnatives in trout streams and mainstem rivers Barriers to prevent trout invasions Flood releases to hamper nonnatives	Control of the water hyacinth. Successful eradication of some fish populations in isolated lakes Anglers directed to kill nonnative fishes captured

[a] Excludes 10 species that migrate to the ocean, like salmon and eel.
[b] CR-includes Endangered and Threatened species under the ESA; IP-includes IUCN Threatened species.

rian rivers have high flows during the autumn-winter rainy season. Both regions suffer many human pressures and have many impoundments on rivers, which is the major source of habitat alteration. For example, on the Iberian Peninsula, >1,200 large reservoirs in Spain alone control about 40% of mean annual flow. Channelization, water abstraction, and water pollution are human stressors of the next greatest importance.

8.5.2. Native and nonnative fish species

The Iberian Peninsula has about 50% more native species than the Colorado River Basin (Table 8.1), and a high proportion in both regions evolved in these basins and are found only there (i.e. 69-80% are endemic). Both regions were long isolated and not covered by continental glaciers, allowing evolution of many endemic fish species. Unfortunately, most of these native species, between 57 and 96%, are imperiled and many have little legal protection. Protection is generally stronger in the Colorado River Basin, with most imperiled species having formal recovery plans under the U.S. Endangered Species Act (ESA), compared to the Iberian Peninsula where almost none have recovery plans yet.

In both regions nonnative species were introduced in the past mainly as sport fish, often by regional governments. Some were released as bait minnows by anglers, especially in the Colorado River Basin, and others were introduced inadvertently from aquaculture, especially in the Iberian Peninsula. A few in each region were unwanted aquarium fishes.

Large nonnative predators like largemouth bass (both regions), northern pike (CR), and European catfish (IP) are depleting native fishes in both regions. However, even small fishes such as mosquitofish (both regions) and small cyprinids like red shiner *(Cyprinella lutrensis;* CR) are capable of preying on larvae of native fishes and thereby depleting them or causing local extinctions. Some nonnatives hybridize with native species, like nonnative trout and suckers in the Colorado River Basin, and invasive cyprinids in the Iberian Peninsula.

Successful non-native species in both regions are well adapted to the newly created reservoir habitats, and the altered flow regimes downstream. The non-native species in the Colorado River Basin and the Iberian Peninsula fill many more ecological niches than the restricted set of niches filled by natives, many of which were well adapted to the fluctuating flow regime (e.g. Olden et al. 2006). In contrast, the successful nonnative species are generalist feeders adapted to warm, slow-moving water, and are weaker swimmers that do not require flowing water and coarse substratum for spawning. The abundance and rate of spread of non-native species has been highest either in reservoirs, or among river fishes that overlap little in life-history traits with the natives.

8.5.3. Prevention and control of invasions

There have been relatively few efforts to prevent invasions in either region, and these have occurred only recently. Fisheries management agencies in the Colorado River Basin are no longer stocking most nonnative fishes, although nonnative trout stocking continues in headwater tributaries and lakes of both regions. Some US states in the Colorado River Basin prevent or restrict the use of nonnative bait minnows for angling, but in both regions illegal introductions by anglers are a major source of new introductions (see Box 8.1). There are no policies in either region to prevent release of unwanted aquarium fish. In the Iberian Peninsula, legislation in December 2011 defined a "black list" of many plant and animal species which cannot be held, sold, or transported within Spain, which could reduce the number of new introductions.

Control of nonnative aquatic species is difficult in any region, and can be accomplished only at the local scale. Nonnative trout are removed from individual Colorado River headwater tributaries, and barriers are often used to prevent nonnative trout from invading upstream (Fausch et al. 2009). Com-

Figure 8.5: *The Colorado River has a high proportion of endemic native fish species, but now two thirds of fish species are nonnative*

plete eradication of nonnative trout is possible only in some small streams, so in most, ongoing removal is required using electricity to capture the fish (Peterson et al. 2008). Releases from reservoirs to mimic natural floods are used in the upper Colorado River to hamper nonnative fish, but monitoring in a major tributary has shown that these floods benefit nonnative species as much as native ones. Large-scale removals of nonnative pike and bass are ongoing in another major upper Colorado River tributary, and do benefit native fish, but represent an expensive long-term management action to prevent their extinction. In the Iberian Peninsula, most funds are spent to control and prevent the further spread of zebra mussel and water hyacinth *(Eichhornia crassipes)* in specific basins. Zebra mussels, common carp, and brook trout have been eradicated in a few lakes, suggesting that other small closed ecosystems might be restored.

Overall, both regions have high proportions of endemic fish species, but are also highly invaded by nonnative species that threaten the native ones. Reservoirs, and the anglers who fish in them, are major sources of new invasions, and these habitats favor nonnative species. Prevention efforts have been too few and too late, and control efforts can generally be effective only at local scales. As for most regions, many more fishes are available worldwide to invade, so invasions will doubtless continue, although perhaps at a reduced pace given the distances involved.

8.6. Implications for conservation

The basic principles about aquatic invasions described above, and the more complex effects caused by other stressors, lead to important implications for conservation of aquatic biota in rivers. Here we describe seven important implications, as statements followed by an explanation.

Nonnative species are here to stay. Once nonnative fauna or flora species establish reproducing populations and spread, it is often impossible or very costly to remove them (Cucherousset and Olden 2011). For example, invasions by nonnative trout, even in streams only 3 m wide and 3 to 5 km long, are difficult and expensive to remove using fish toxicants like rotenone. Often, ponds created by beaver *(Castor canadensis)* or groundwater seeps provide refuges where a few nonnative fish survive. Carefully planned projects can be successful in small streams with relatively simple habitats, but virtually none could eradicate nonnative fishes or invertebrates in rivers at least 10 m wide. Moreover, it may be more cost effective to prevent further introductions into new waters, such as illegal introductions by anglers or aquarists (see below).

An ounce of prevention is worth a pound of cure. This old adage is especially true for invasions, and argues for much more focus on wise policies of preventing invasions than attempts to eradicate them after they establish and spread. Willful introductions by fisheries management agencies are less often a major vector of introduced species now than in the past, and many now prevent use of nonnative fishes as bait in specific watersheds. Likewise, recent research on the potential vectors of ballast water and aquarium fishes show that sharply reducing these vectors is technically possible. However, methods to reduce the growing trend of introductions of invasive piscivorous or forage fishes by anglers are in desperate need of attention by social scientists and fisheries biologists (Johnson et al. 2009; Box 8.1). In addition, early intervention using all means available to remove an invasion that is limited to a small area is far more effective than attempts to remove, control, or adapt to invasions after they have spread (Simberloff 2003).

It is worth closing the barn door after the first batch of horses is gone. Recent research and synthesis (Simberloff 2009) indicate that propagule pressure, the number and frequency of organisms introduced, is one of the most important factors driving invasion success. Managers may assume that once a species has arrived in a new location, there is little use in preventing future arrivals and introductions. However, many species may require multiple introductions, or a minimum number of propagules introduced at one time, to successfully overcome environmental or biotic limits and become established. Therefore, limiting further introductions can be highly effective at preventing invasions.

Protecting natural habitat and disturbance regimes may favor native species over nonnatives. Changing environmental factors is a powerful force that can shape groups of species in habitats. For example, the Natural Flow Regime Paradigm (Poff et al. 1997) holds that native species are strongly hampered when water abstraction alters natural flow and flood disturbance regimes, providing "niche opportunities" for nonnative species to exploit, thereby leading to invasions (see Olden et al. 2006). The converse has also been proposed, that restoring natural flow regimes may help reduce nonnative species abundance. There is some evidence to support this claim, but in other cases both natives and nonnatives can flourish under natural flow regimes (Merritt and Poff 2010). More work will be needed to test this important assertion, because many management schemes are based on this theory.

Dammed if you do, but perhaps damned either way. Managers often consider preventing upstream invasions into headwater streams using natural or artificial barriers to conserve native species. Many of these are simply road culverts or other human-made structures that have already prevented such invasions. However, isolating fish or invertebrate populations in short headwater fragments can also hasten their extinction, either because the organisms lack all essential habitat needed to persist (especially after natural disturbances like fires, floods, freezing, or drying) or because populations are currently supplemented by immigrants from downstream. This tradeoff of using barriers to prevent invasions has been identified (Fausch et al. 2009), and methods have been developed to optimize it for native stream salmonids in the western USA. However, much more research will be needed for other species and regions to determine the lengths of stream fragments needed to support populations of native species into the future.

Preventing invasions in the first place is the most cost-effective management option, but control of invasions after they spread is often given the highest priority

Nonnative species can have far-reaching effects, even beyond the stream. Nonnative organisms may rapidly spread long distances, especially because flowing waters transport propagules and adults downstream. Paradoxically, effects of nonnatives can also be transferred to adjacent habitats across the terrestrial-aquatic boundary. For example, riparian invasions by nonnative tamarisk alter light and leaf inputs to streams, thereby altering organic matter dynamics and changing invertebrate and fish assemblages. Conversely, nonnative trout invasions can cause cascading effects within stream food webs that increase algae, and reduce the flux of emerging adults of aquatic insects that are major components of the energy budgets for riparian birds, bats, lizards, and spiders (Box 8.2; Baxter et al. 2004; Benjamin et al. 2011). Managers of either terrestrial or aquatic habitats often do not consider the effects of such invasions on the adjacent, tightly-linked ecosystem, which can confound efforts at restoration.

Look before you leap, to control. When eradication is not possible, managers often consider long-term control measures to keep nonnative species from potentially replacing native species. After invasions have spread, such efforts are an expensive and ongoing cost. Long-term control may be an important option for species of high economic or conservation value, but such efforts are worth analyzing carefully, to optimize efforts. Peterson et al. (2008) linked population models for a native and invasive trout, including the effects of the nonnative on the native, to estimate the efficacy of different scenarios of frequency and magnitude of mechanical removal by electrofishing on persistence of the native trout population. Continuous annual control was not as cost-effective as successive 2-3 year periods of control. This pulsed control allowed survival of a cohort of juvenile native trout through the first two years of life, the only period when the relatively long-lived native trout were vulnerable to competition or predation from the nonnative.

8.7. What should be our priorities?

Managers often have limited time, expertise, and funding, and many consider that they have few options for addressing invasions by nonnative organisms in stream and river ecosystems (Fausch et al. 2009). Many seek advice from scientists about where to start. In contrast, scientists often study questions that are interesting or personally rewarding, sometimes bypassing less interesting problems that, nevertheless, could be more effective at reducing or controlling invasions. Here, we seek to arrange management and research actions in order of importance for stemming the tide of invasions (Table 8.2).

The most important research involves profiling likely invaders at the arrival phase, to identify those that are likely to do the most damage. However, many

Table 8.2: *A list of potential management and research actions to reduce invasions of nonnative biota into river ecosystems, in order of priority from the most to least important*

Action	Explanation	Relevant references[a]
Profile and prevent	Estimate which species are most likely to arrive, by which vector and where. Assess which species are most likely to do damage if they spread	Kolar and Lodge 2002 Alcaraz et al. 2005 Vila-Gispert et al. 2005 Garcia-Berthou et al. 2005
Educate public, and limit vectors	Educate the public about preventing arrival or spread of dangerous invaders, and develop methods to reduce risk of human or natural spread	Vander Zanden and Olden 2008 Strecker et al. 2011
Reduce propagule pressure	Even after species have arrived, seek ways to reduce propagule pressure, a primary driver of nonnative species establishment and spread	Von Holle and Simberloff 2005 Simberloff 2009
Manage habitat and flow regimes to favor native species	Habitat change is a powerful force that hampers native species survival and provides niche opportunities for invasions. The converse, that restoring natural flow or other disturbance regimes will reduce nonnative species invasions, is not always true but deserves more study	Poff et al. 1997 Lytle and Poff 2004 Marks et al. 2010 Merritt and Poff 2010 Hermoso et al. 2011
Consider tradeoffs in social values and management options	The public may value some invaders while considering others noxious pests. This perception may change with more information, which deserves research	Fausch et al. 2006
Understand establishment and spread	Once invaders have arrived, understanding what allows them to establish and spread may allow the development of management actions to limit these stages, or suggest stronger policies to reduce propagule pressure	Fausch 2007 Garcia-Berthou 2007
Eradicate or control	Eradication may be possible at the early stages of invasion when the spatial extent is limited, and should be pursued with all means possible. Control is often difficult and entails large and long-term costs	Simberloff 2003 Peterson et al. 2008 Vander Zanden et al. 2010

[a]See Additional References for some references listed here.

ecologists may prefer to study what allows nonnatives to establish, which we place sixth in priority. Once dangerous invaders have been identified, the second most important work is to educate the public and to seek methods to limit vectors of these species. Both of these topics require interacting with social scientists and regulatory agencies, which many ecologists avoid. In contrast to these high priorities, the action least likely to be effective at preventing or eliminating invasions is direct control of nonnative species, even though this is often the first action considered by many ecologists and managers.

8.8. Conclusions

To conclude, two things are obvious. First, invasive species are a huge issue for river conservation. They deserve more resources, especially to prevent their introduction, but also to measure their ecological impacts and develop better methods of control. Second, many management options are technically possible, but need proper prioritization. Although managers often consider control of invaders their first priority, prevention would be more cost effective in the long run. Perhaps because invasive species are underwater and so not readily visible and particularly because they are living organisms, the public tends to appreciate this problem less than others affecting rivers. Therefore, the first step needed to improve management is effective communication and public awareness at many different levels.

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Chapter 9

Between the Land and the River: River Conservation and the Riparian Zone

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The riparian zone is the transition between the land area of the catchment and the river channel. Riparian zones are areas with unique biodiversity and extremely important ecological functions, but they are currently threatened by increased human pressures. The ability of near-stream land to buffer the river channel is unique, and opportunities to rehabilitate these areas could benefit the whole river basin.

9.1. Why consider the riparian zone?

This chapter is about riparian land, the area bordering the river channel. Strictly speaking, the riparian zone includes only vegetation along the bed and banks of the river channel but in recent years the definition has extended to include the wider strip of land alongside the channel. Riparian zones are ecological boundaries, or ecotones, separating terrestrial and aquatic ecosystems. In headwater valleys, the riparian zone will be narrow, just a few metres wide at most, but lower down the river network, it can be very much wider, tens of kilometres wide on the largest rivers. Often the riparian zone is taken to be synonymous with the floodplain, the area of low-lying ground adjacent to a river, formed mainly of river sediments and subject to regular flooding. Floodplains cover an area of the order of millions of square kilometers worldwide (Tockner and Stanford 2002) and, thus, are quantitatively very important.

The idea of an *ecotone* was first proposed by Clements (1905) to denote the junction between two distinct biological communities: "*a zone of transition between adjacent ecological systems, ecotones have a set of characteristics uniquely defined by... the strength of interaction between the adjacent ecological systems*" (Holland 1988). Riparian zones fit perfectly with this definition, since they can play varied roles, shifting in time from a character that is reflective of the upland, terrestrial system, to one that may be more like the river (i.e. a conveyance system). Their position between rivers and uplands means riparian zones are effectively boundaries that can be described in terms of their permeability, width, gradient and so forth (e.g. Strayer et al. 2003). The overall biodiversity of riparian areas is extremely high, resulting from the unique combination of an ecotone between two contrasting ecosystems, from fertile soils and from the natural regime of floods and droughts (Naiman et al. 2005; Figure 9.1). In addition to the species characteristic of the interface between water and land, riparian areas often receive visitor species from the surrounding landscape, that go there to make use of the available resources or, more often, that use riparian areas as a corridor, given their spatial configuration. At the same time, riparian zones are among the most threatened ecosystems in the world, as humans also seek access to the river margins and convert fertile riparian soils for agriculture. For instance, in densely populated areas of Europe and Asia, between 60% and 99% of the entire river corridor has been converted to agricultural or urban areas (Tockner and Stanford 2002).

Figure 9.1: *A flooded* varzea, *the name given to riparian forests in the Amazon basin. These forests are extremely productive and diverse*

Figure 9.2:
Riparian zones provide many ecosystem services (Table 9.1). Here, on the River Test in southern England, the riverside hay meadows provide grazing for animals and a rich habitat for flora and fauna. There is also access for anglers; this stretch of river was the birthplace of fly fishing

There is an intuitive assumption that the condition of the stream and the condition of the riparian zone are intimately linked. In general there is agreement that, for the good of the in-stream habitat, near-stream land should be maintained in as natural a state as possible. Until recently, riparian zones were thought of mainly as productive farmland or good sources of timber but their distinctive biota and apparent ability to protect the stream environment has prompted renewed interest in their broader ecological function. A good deal of specialist research has focused on the use of riparian land as an effective means of preventing diffuse pollution from farmland from reaching the river channel. As Bren (1993) points out, near-stream land is popular for all sorts of human activities – from farmland to recreation – so there is bound to be disagreement about the best use of this land (Figure 9.2).

Four primary ecological functions can be identified in the riparian zone, with a fifth to emphasise human use of riparian land (de Groot et al. 2002; de Groot 2006):

Regulation functions. These arise where stable ecosystems are able to buffer the impact of extreme hazards and provide some stability to the natural environment. They include air quality, climate, river flow, soil erosion, water purification, disease and pest control, and pollination (Table 9.1). By definition, we can expect the floodplain to be flooded on a regular basis, every year or two on average. In terms of flood protection, it is now realised that floodplains provide

Table 9.1: *Functions of natural and semi-natural riparian ecosystems and their translation to ecosystem services*

	Functions	Ecosystem processes and components	Potential ecosystem services
		Regulation function: maintenance of essential ecosystem processes	
1	Gas regulation	Role of riparian wetlands in gas exchange with the atmosphere via biogeochemical cycling (e.g. CO_2, N_2O, CH_4)	Improved air quality, prevention of climate change
2	Climate regulation	Influence of land cover on boundary layer climate	Favourable conditions for biota
3	Hazard protection	Storage of flood waters, attenuating the flood wave downstream	Downstream flood protection
4	Nutrient and pollutants regulation	Biogeochemical cycling in riparian soils; processing of pollutants derived from the surrounding terrestrial ecosystem	Protection of water resources
5	Soil protection	Role of vegetation cover in preventing soil erosion	Protection of water resources
6	Pollination	Role of biota in movement of floral gametes	Pollination of crops and wild species
		Production function: provision of natural resources	
7	Food	Conversion of solar energy into edible plants and animals	Hunting, gathering of fish, game, fruits
8	Raw materials	Conversion of solar energy into biomass for human uses	Timber for building, fuel, fodder
9	Genetic resources	Genetic material in wild plants and animals	Medicines, drugs, pharmaceuticals
10	Ornamental resources	Growth of biota with potential ornamental use	Resources for fashion, handicraft, jewellery, decoration
		Habitat function: provision of habitat for wild plants and animals	
11	Refugium function	Suitable living space for wild plant and animal species (including migrants)	Maintenance of biological and genetic diversity
12	Nursery function	Suitable reproduction habitat, both riparian and in-stream	Maintenance of commercially harvested species
		Information function: providing opportunities for cultural experiences	
13	Aesthetic experiences	Attractive landscape features with potential cultural and artistic value	Enjoyment of scenery, use of landscape in art
14	Recreation and tourism	Maintenance of landscape variety	Leisure pursuits (e.g. walking, angling)
15	Spiritual and historic information	Preservation of historic artefacts	Heritage value of natural and human features, eco-tourism

Table 9.1 *(cont.)*

	Functions	Ecosystem processes and components	Potential ecosystem services
16	Science and education	Variety in natural ecosystems with scientific or educational value	Use of riparian zones for out-of-classroom education and scientific research
	Carrier function: providing a suitable foundation for human activities and infrastructure		
17	Habitation	Providing a suitable location for human settlement and transport infrastructure including the provision of aggregate for the construction industry and locations for waste disposal	Living space; mining
18	Cultivation	Providing a suitable location for farming, commercial forestry and bio-fuels	Crop production

Source: Adapted from de Groot (2006).

important storage of flood waters. Without this (for example, if the floodplain is "protected" by levees), flood water moves quickly on, often to flood the next settlement downstream. In ecosystem terms, flood storage is a regulation function but the riparian zone fulfils other regulation functions too: for example, nutrient export may be reduced and local climate modulated. Later, we focus on biogeochemical cycling in riparian soils (e.g. nitrate, phosphate, carbon) and its dual influence on nutrient loss from the catchment area and gas exchange (e.g. nitrous oxide) with the overlying atmosphere.

Production functions. These ecosystem functions underpin the provision of natural biotic resources. These include wild plants and animals as sources of food (e.g. fish, game) and genetic resources – in some countries riparian ecosystems are an important source of medicinal compounds. Riparian land may also provide ornamental resources, items for fashion and handicraft such as wood, jewellery and flowers, and may be a rich source of timber, fuel and food for some people.

Habitat functions. In its natural state, a riparian wetland includes many specialised habitats. If riparian wetlands are intact along the entire river channel, they can also provide an important pathway for species migration.

Information functions. Natural ecosystems provide essential cultural services, contributing to human health and well-being by providing opportunities for reflection, spiritual enrichment, cognitive development, recreation and aesthetic experience. This includes the preservation of elements of landscape history. Access to floodplains for walkers and anglers is important to many people.

Carrier functions. Most human activities (e.g. housing, transport) require space and a suitable foundation to support the associated infrastructure; most of these activites involve complete destruction of the original ecosystem. Floodplains have always provided humans with living space and today, in towns and cities, they continue to provide flat ground for housing, industry and transport. This space is so valuable in monetary terms that its fundamental nature gets forgotten, until the next flood that is. In many places, floodplains are also a convenient source of aggregate for the construction industry, an activity that again conflicts directly with habitat conservation (Chapter 3). Farming can also be included in this category, given that farmland is clearly different from the natural ecosystem it has replaced. The soil is often very fertile, particularly when the water table has been lowered by land drainage. Intensive farming for the production of food, fibre, timber and, increasingly, bio-fuel is likely to compete with habitat functions: a more varied landscape which includes woodland and wetlands will have much higher biodiversity than arable land. Traditional low-intensity farming methods such as hay meadows are valued for their rich flora, and farmers may be paid to conserve them.

It has been estimated that these functions of floodplains are responsible for more than 25% of all the terrestrial ecosystem services, despite floodplains covering only 1.4% of the land surface area (Tockner and Stanford 2002).

9.2. Hydrology of the riparian zone

Given their location (adjacent to the river channel) and topography (often a wide, flat area), riparian zones are more often than not likely to have high water tables, even if the substrate is permeable. Very low gradients across the floodplain help to sustain waterlogged conditions, especially where the floodplain is wide or the alluvial sediments are of low permeability. Often, the riparian zone is so poorly drained that peat deposits have accumulated, adding to its poorly drained condition still further. Inputs of water to the riparian zone can originate both from the catchment area adjacent to the riparian zone and from the river channel, as well as from precipitation (Box 9.1). In headwater valleys, the main direction of water movement will be from land to river channel but further downstream there is more of a balance between these sources of water.

9.2.1. Flow paths and the residence time of water within the riparian zone

In headwater tributaries the riparian zone may be very narrow or non-existent, so that the opportunity for the riparian zone to buffer the impact of terrestrial runoff will be minimal. In the middle sections of the stream network, the presence of floodplains provides the potential for buffering runoff from the catchment as well as providing storage for flood waters. In lowland reaches there may be very wide floodplains with no connectivity between "upslope" areas and the river.

In *headwater catchments* slopes are intimately coupled to streams; the predominant direction of water movement is towards the stream (Burt et al. 2010). There

Box 9.1

The water balance of the riparian zone

The water balance of the riparian zone may be defined in terms of the inputs and outputs to the area (Burt et al. 2010):

Inputs

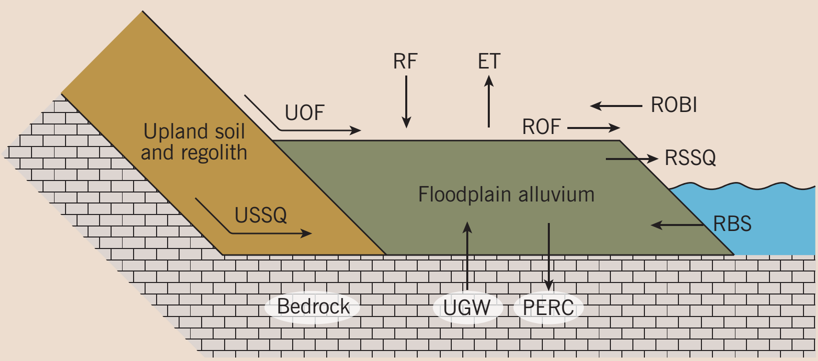
a. Overland flow from the terrestrial ecosystem upslope (UOF)
b. Subsurface flow from upslope (USSQ)
c. Precipitation directly on to the riparian zone (RF)
d. Groundwater discharge from local aquifers into the riparian zone (UGW)
e. Seepage from the river channel through the bank (RBS)
f. Overbank flooding from the river to the floodplain surface (ROBI)

Outputs

a. Overland flow from the riparian zone to the river (ROF)
b. Subsurface discharge from the riparian zone to the river (RSSQ)
c. Evaporation from the riparian zone (ET)
d. Percolation from the riparian zone into aquifers below (PERC)

Any difference between input and output must, by definition, involve a change of water storage within the riparian zone (ΔS). The water balance may therefore be expressed as follows:

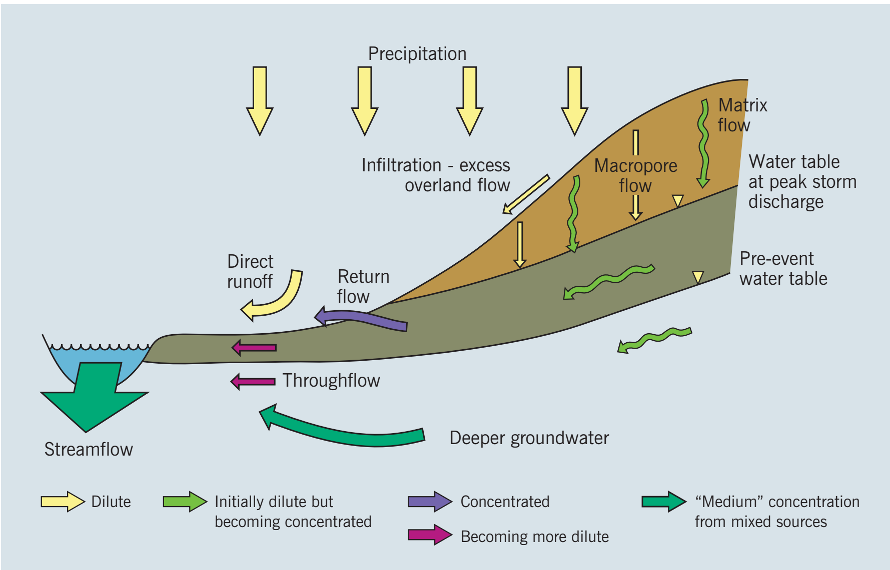
$$UOF + USSQ + RF + UGW + RBS + ROBI - ROF - RSSQ - RET - PERC \pm \Delta S = 0$$



may or may not be a narrow riparian zone with the potential to provide some protection for aquatic ecosystems, but hydrological conditions in the near-stream zone are predominantly controlled by inflows from upslope. Figure 9.3 provides a schematic representation of hillslope flow processes and associated nitrate transport. It shows how water flowing rapidly across upslope soils remains dilute whereas water flowing more slowly through soil and bedrock is much more likely to become concentrated. In relation to nitrate, the riparian zone may protect the stream: waterlogged soils favour anaerobic processes like denitrification, with nitrate being reduced to nitrous oxide or dinitrogen gas and thereby permanently removed from the river basin. However, the same conditions may favour release of other nutrients e.g. phosphate, so that riparian zones may not buffer all pollutants to the same degree. Surface runoff on farmland may erode soil; this may be deposited in the riparian zone, depending on its width and the type of vegetation cover found there.

In the *middle reaches* of a river basin floodplains are wider and there can be inputs to the riparian zone from both hillslopes and the river channel (see Box 9.1). Bank storage is an important process during flood events, both seepage through the bank and overbank flooding from the river to the floodplain surface. Hillslope discharge to the riparian zone dominates during non-flood periods. In temperate zones, the main emphasis has been on buffering as water moves from upslope areas (usually farmland), across the riparian zone to the

Figure 9.3: *Hillslope runoff pathways and associated nitrate transport*



Source: Burt and Pinay (2005).

stream; there is a relatively small buffering capacity for water moving out of the channel during floods. However, in semi-arid areas, water movement out of the channel is a much more important source of water compared to temperate areas (Harms and Grimm 2008).

In *large river basins* the floodplain becomes an important source of runoff in its own right and there is little influence from the surrounding catchment area. Drainage of the riparian zone to allow more intensive agriculture encourages subsurface flow and may increase nutrient leaching as a result. Water draining through the soil can by-pass most of the riparian zone via ditches and drains, much reducing the opportunity for buffering processes to operate. For example, it is thought that rising nitrate concentrations in many rivers in the UK in the 1960s and 1970s were in part caused by extensive land drainage programmes at that time, much of which involved drainage of floodplains (Burt et al. 2008). This was compounded by the fact that land use changed from low-intensity grazing to high-intensity arable farming, with ploughing annually and high rates of fertiliser application.

9.2.3. Hydrological variability and disturbance as drivers of change in the riparian zone

Fluvial processes shape the form of river channels over the long term (decades to centuries) through processes of erosion and deposition (Chapter 3). The familiar example of a migrating meander illustrates how these slower geomorphic processes influence riparian zones: on the outside edge of the meander, trees succomb to the flow even as new substrate for seedlings is deposited on the opposite, aggrading the bank. Successional processes of vegetation growth integrate with the dynamic change in river channel form, creating complex patterns in substrate (soil, sediment) and biota upon which biogeochemical processes play out.

On ecological time scales, individual floods are disturbances that contribute to the geomorphic landscape evolution, but also are important drivers of change in riparian structure and function. Floods can uproot trees, carve out river banks, deposit thick layers of sediment in some areas and scour others. They will generally produce a rise in water table within floodplains and may displace riparian groundwater, causing pre-event soil water to mix with river water. Rising water tables can promote soil microbial activity by alleviating water limitation. The channel and the riparian zone differ in their resilience to flood disturbance (Fisher et al. 1998); in general, riparian zones are less likely to be altered by all but the largest floods compared to the stream channel which will be regularly disturbed. On the other hand, because they often are dominated by long-lived organisms (trees), when flood destruction occurs, they will re-establish more slowly.

Drying also is a disturbance to riparian zones; indeed, the prevalence of seasonal drying may limit the extent of riparian zones to larger streams, particularly in arid, semi-arid, or Mediterranean climates. Stream drying during regional droughts can decimate riparian forests when the water table falls below the reach of riparian vegetation during long periods. It follows that any propensity for climate change towards warmer and drier conditions, i.e. increasing evaporation losses relative to rainfall input, will pose a threat to riparian habitats.

The particular pattern of seasonality in flow, differences between peak and low flow, timing and magnitude of floods, and duration of extreme low flows comprises the hydrological regime. Hydrological regimes differ among climatic regions (Chapter 2) and it is important to understand not just the impact of an individual disturbance but of the entire regime. In the arid South West of the USA, mineralization of organic matter is a major source of available nitrogen, subsidized by input of nitrogen from floods. Baseflow inputs are most likely removed by rapid denitrification at the stream-riparian edge, while higher rates of flood supply exceed the capacity of this “filter” (Schade et al. 2002). Year-to-year hydrological variability is very high and results in multi-year differences in the abundance of a shrub, *Baccharis salicifolia*, that colonizes the parafluvial zone (nearest-stream portion of the riparian zone). Because *B. salicifolia* roots alter subsurface organic matter content and flow patterns, these difference between years translate to strong impacts on nitrogen biogeochemistry.

A complex set of interactions governs the hydrological disturbance regime in any catchment. Floods are not easily predicted simply from rainfall amount and intensity; the permeability of soils, antecedent conditions (how long since it last rained), soil and vegetation type, temperature, and so forth all contribute (Chapter 2). Thus, it is clear that hydrological regimes are likely to be altered under global climate change, although we are far from being able to generate predictions with high confidence. With changing hydrological regimes, we expect to see changes in the character and biogeochemical dynamics of riparian ecosystems.

9.3. Biogeochemical cycling in the riparian zone

Riparian zones have long been under human pressure because of conflicting interests associated with the use of near-stream land. The fundamental role of these wetlands in the functioning of river ecosystems has been ignored until relatively recently (Burt et al. 2010). Even though ecologists have been interested for decades in spatial transitions from one biological community to another, and how their proximity affects the functioning of each zone, science and management have been disconnected (Grimm et al. 2003). The importance of the riparian zone ecotone as a "buffer" against high sediment and nutrient (nitrogen and phosphorus) fluxes from land to the sea via riverine transport has been recognized in terms of diffuse pollution control (Peterjohn and Correll 1984).

9.3.1. The riparian zone as conduit

Riparian zones, the land areas bordering the river channel, have unique biodiversity and extremely important ecological functions

Given their location alongside rivers, during flood events riparian zones receive large amounts of dissolved and particulate organic matter and nutrients from upstream. In headwater locations, riparian zones are subject to large subsurface nitrate inputs from the adjacent uplands (Peterjohn and Correll 1984), while in larger rivers, significant amounts of sediment, organic matter and nutrients are deposited during overbank flood events. River floodplains are recognized as important storage sites for sediments and associated nutrients mobilized from upstream catchments during floods (Walling and He 1998). The recycling and storage of sediment deposits in floodplains are largely depend on the hydrological connectivity between the river and its floodplain, i.e. existence of side channels and oxbows, as well as of the magnitude, frequency and duration of floods. Collectively, these factors create a mosaic of geomorphic surfaces that influence the spatial pattern and successional development (series of vegetation community from pioneer grass, to soft and hard wood) of riparian vegetation (Salo et al. 1986). The fluxes of matter mediated via surface connectivity have the potential to control gaseous nitrogen loss via denitrification by controlling the rate of nitrate delivery. This has been shown for pools in the Danube River (Welti et al. 2012) and in other smaller European floodplains (Pinay et al. 2007). In riparian zones and floodplains well connected to the river, the pattern of surface and subsurface flow provides large potential for nitrogen retention and removal which contributes to reduction of natural diffuse pollution (Burt and Pinay 2005).

The high productivity measured in floodplains is mainly a function of the abundant matter supplied by the drainage basin as well as the co-existence of aerated (oxic) and non-aerated (anoxic, reduced) conditions in its soils and sediments (Brinson et al. 1984). In many parts of the world, floodplains sustain high food production for the local population. For instance, flood events in a given year increase the fish yield the following year in various large rivers such as the Danube, the Kafue, the Niger and the Shire rivers (Welcomme 1995). Sediment and nutrient deposits on the Ganges and the Brahmaputra floodplain mean soils can sustain up to three rice crops a year in Bangladesh.

9.3.2. The riparian zone as a barrier

The use of natural buffer zones to protect fresh water from pollution has attracted considerable interest. It is now recognized that riparian zones along streams can mitigate diffuse pollution by nitrate input from upland areas. Two processes are involved in this regulation: plant uptake, which provides temporary storage, and denitrification, which represents a permanent sink for nitrogen since nitrate ultimately is transformed to a gaseous form and lost from the river ecosystem completely (see Haycock et al. 1997; and Burt et al. 2010 for reviews).

There is an intuitive assumption that the condition of the stream and the condition of the riparian zone are intimately linked

Efficiency of nitrogen cycling and retention, the processes which contribute to diffuse pollution control in river ecosystems, is correlated with the length of contact between water and sediment in stream or between wetland and upland. This positive relationship occurs both in the main channel itself and in the riparian and floodplain zones (Hill 1979; Jones and Holmes 1996; Valett et al. 1996). The duration of contact between water and these substrates controls the biological use and thereby the total amount of nitrogen processed. The frequency, duration, timing and intensity of floods also directly affect nitrogen cycling in alluvial soils by controlling the period during which soils will be saturated with water and therefore will lack aeration. This soil saturation with water can result from flooding but may simply reflect the slow rate of drainage across the flat riparian zone. Flooding duration is controlled by local topography: low areas are flooded more often and for longer than higher ones. Biogeochemical processes involved in nitrogen cycling are sensitive to whether the soil contains free oxygen or not (Hefting et al. 2004). For example, organic nitrogen can be transformed into ammonia by both aerobic and anaerobic ammonification processes in oxic or anoxic conditions respectively, whereas the nitrification process, which requires free oxygen in the environment, can only occur in aerated soils or sediments. As a consequence, under permanently anoxic conditions, mineralisation of organic nitrogen results in the accumulation of ammonium. Other processes, such as nitrate dissimilation or denitrification, are anaerobic and require saturated soils to operate. Therefore, the end products of nitrogen cycling in riparian soils are controlled by the moisture regime (i.e. water table level), with important implications for floodplain productivity and management.

It is important to underline that the capacity of riparian zones to retain and remove nitrogen does not apply to other types of pollutants. It is especially clear, for instance, that the role of riparian forests in controlling phosphorus pollution has been often overestimated. Phosphorus is mainly transported by surface flow and its permanent removal from riparian wetlands can only be achieved by plant harvesting since it does not have any gaseous form. Phosphorus is somewhat less mobile than nitrate, forming insoluble complexes, but under anoxic conditions phosphorus goes back into solution. Thus, riparian zones may become sources of soluble phosphorus for the adjacent stream under flooded conditions. This limits their role on phosphorus flux control (Uusi-Kamppa et al. 1997).

9.3.3. Hot spots and connectivity at the landscape scale

Riparian zones represent an important interface between the terrestrial and aquatic environments and can exert significant controls on water quality. They are typically areas of topographic convergence with high upslope contributing area and low slope which promote the development of near-surface saturation and enhanced denitrification. In addition, the combination of reduced slope and increased heterogeneity due to the presence of trees and rough grass can enhance deposition of soil eroded in adjacent fields and the removal of associated organic matter and nitrogen from runoff (Burt and Pinay 2005).

Nevertheless, factors accounting for the pollution retention capacity of riparian zones are diverse, and the performance of a buffer zone within a catchment is difficult to predict (Haycock et al. 1997). Indeed, the transfer of nitrogen within the drainage basin and its transformation within riparian zones varies widely in response to local environmental conditions. For instance, Pinay et al. (1998) examined the buffering capacities of different riparian vegetation (natural riparian forest, 3- and 15-year-old poplar plantations, and a wet meadow) on non-point source nitrogen pollution along a 7th-order reach of the Garonne River in south west France. They found that the role of riparian zones was marginal. In an urban study, Roach and Grimm (2011) compared denitrification among habitats of a constructed stream-pond-floodplain complex in south western USA, and found that denitrification in grassy floodplains that were periodically inundated or irrigated removed nearly all of the nitrogen added by fertilisation, but that denitrification in the ponds was limited by nitrate diffusion through the sediment and in the streams by a small areal extent. This designed floodplain thus provided nitrogen removal service within the larger urban landscape. In a pan-European study evaluating the role of small forested and meadow riparian zones, Sabater et al. (2003) found that the rates of biological uptake and denitrification of nitrogen were controlled by local hydrological conditions and nitrate load rather than by broad differences in climate among sites. The large variability of nitrate export rates from small headwater basins is a sure sign that nitrate retention processes are very active at some sites but completely absent in others (Burt and Pinay 2005). These two last studies point to the high degree of variability among sites and a limited predictive capacity based upon broad-scale drivers.

Given the high heterogeneity at the local scale (topography, soil, vegetation cover, etc.), it is difficult to extrapolate site specific *in situ* evaluation of nitrogen buffering capacity of riparian zones at larger scales, i.e. 1 to 100 km^2. This intermediate catchment size is also the scale where models linking percentage of land use to nutrient fluxes tend to fail (Strayer et al. 2003). However, this is an important management scale where socio-economical drivers such as crop production and landscape aesthetics meet. An alternative approach to tackling this scaling issue could be to consider that riparian zones represent a particular type of biogeochemical hot spot where hydrological flow paths converge with high concentrations of substrates (such as soil carbon and nitrogen) essential for microorganisms. These "coupled" solutes are transported to the riparian zones which show disproportionately high reaction rates relative to the surrounding matrix (McClain et al. 2003). Therefore, evaluation of nitrogen retention and removal at the drainage basin level could be done by considering the likelihood of a given land use and land cover arrangement hosting biogeochemical hot spots.

9.3.4. Contrasting cases: temperate, arid, and arctic riparian zones

The previous overviews mainly describe general hydrological and biogeochemical conditions that typify riparian zones of temperate regions. In other regions, seasonality of the hydrological cycle and ecosystem processes yields patterns in riparian biogeochemistry that contrast from the general, moderately moist ("mesic") model. Here, we discuss riparian zones that differ from this general model. Patterns observed in these special cases may also pertain to temperate riparian zones under conditions that differ from normal, including drought or urbanization.

Drylands. Temperate rivers tend to receive water from the aquifers and from multiple subsurface sources, and therefore, are called "gaining" rivers, as the discharge they transport tends to increase downstream. In contrast, rivers in dry areas are called "losing" rivers, as they tend to lose water to local aquifers and to the riparian zone. The direction of this flow has consequences for both hydrology and biogeochemistry. Riparian zones along losing reaches have deeper groundwater tables than those along gaining reaches, and surface flow is often intermittent or ephemeral. Overall, water availability is much lower in the riparian zones adjacent to losing reaches. As a consequence of the scarcity of water, riparian vegetation is less dense and rates of soil microbial activity are water limited; thus the capacity for nutrient retention is much lower in riparian zones along losing compared to gaining reaches (Harms et al. 2009).

Although water is scarce for much of the year in arid regions, large floods occur from time to time. Because the soils, devoid of much vegetation, have low infiltration capacity, the heavy rainfall falling during a storm quickly reaches the stream. This results in inputs of water from up-basin tributaries (often ephemeral washes), overbank floods that inundate the riparian zone, and a rapid rise of the water table. Because soluble materials can build up in soils during long dry periods, inputs to the riparian zone are accompanied by high loads of dissolved and suspended materials from the uplands and "flushing" of solutes derived from riparian soils. Sediments may be physically entrained or trapped by riparian biota, whereas increased water availability combined with increased availability of nutrients can promote biological uptake and removal of carbon and nutrients. However, during very large floods the residence time of water and substrates in riparian zones may be insufficient to allow significant biological activity, and most of the nutrients are exported. Conversely, in locations where there is prolonged inundation, this may also suppress biological uptake due to declining oxygen levels and substrate availability. Thus, the size and timing of water inputs to riparian zones of drylands has strong consequences for biogeochemical activity (Harms and Grimm 2012).

Floodplains are estimated to be responsible for more than 25% of terrestrial ecosystem services despite covering only 1.4% of the land surface area

Permafrost-influenced catchments. Permafrost is ground that remains frozen throughout the year, and is common at high latitudes or high elevations. During summer, the soil surface can thaw (the thawed soil is known as the *active layer*), but the deep soil layers remain frozen. Catchments dominated by permafrost have unique hydrological templates that have consequences for the biogeochemistry of riparian zones. Permafrost restricts deeper percolation of soil water, preventing growth of plant roots and fostering little microbial activity. Water moving from upslope areas via riparian zones to the stream flows through the active layer.

Thaw dynamics play a dominant role in the hydrology and biogeochemistry of permafrost-influenced catchments. Early in the snowmelt period, soil thaw is minimal, and solutes and water in the snowpack are exported from the riparian zone. However, some time later the upper organic soil horizons, which are typically composed of living mosses, begin to thaw, and thus provide strong potential for retention and removal of nutrients. As the soils continue to thaw, flow paths may be disconnected from surface organic horizons, and flow is routed through deeper, mineral soils. These soils may strongly adsorb organic molecules, but provide a weak sink for inorganic solutes. In sum, seasonal patterns in thaw depth and water table elevation in riparian soils contribute to strong seasonality in solute export.

Spatial extent of permafrost and the rate of seasonal thaw of soils respond strongly to the thermal regime. In regions with discontinuous permafrost in the Northern Hemisphere, south- and west-facing catchments tend to have less permafrost. Similarly, where permafrost is continuous, deeper active layers form in catchments that receive greater solar input. Permafrost extent and depth of thaw have consequences for the residence time of water in the riparian subsurface. Water can infiltrate thawed soils, which provide a reservoir for water storage, and the riparian zone contributes more strongly to mitigating peak flows and material fluxes during storms where thaw depth is greater.

Riparian zones in permafrost regions are particularly prone to bank destabilization due to the thawing of ground ice. Bank collapse features are particularly common along larger rivers (Figure 9.4). Once initiated, these features rapidly develop, with stream banks often eroding at rates of metres per year. Formation of thermokarst (hummocky ground formed by thawing of ice-rich permafrost) has dramatic consequences for riparian hydrology and biogeochemistry by removing vegetation from the riparian zone, exposing mineral soil, and enhancing export of sediment and nutrients.

Figure 9.4: *Thaw slumps in permafrost regions can cause extensive and rapid downcutting of stream channels, removing riparian vegetation and exporting riparian soils and sediments downstream*

9.4. Human drivers of change in riparian zones

Although riparian areas are extremely important from the point of view of the biodiversity they host, as well as of the services they offer, they are also among the most threatened areas of the world (Tockner and Stanford 2002). In Europe and North America up to 90% of floodplains are severely modified for agriculture, intensive forestry or urban uses, and riparian habitats are among the most threatened by expansion of human activities. Here we discuss briefly some of the human pressures driving changes in riparian zones.

9.4.1. Hydrological regime

Human activities in any location within a catchment will affect ecological functions and their translation to ecosystem services. In the uplands, groundwater extraction can cause streams and riparian zones to dry out by reducing streamflow and drawing down the water table. When hydrological inputs from the surrounding uplands are lost, the subsurface connection between streams and riparian zones can be reduced and riparian vegetation may no longer have access to a perennial source of water. Dewatering of stream-riparian corridors has occurred extensively in arid regions, and has consequences for plant species richness. Plant species richness declines as flow permanence declines in desert riparian zones; loss of obligate wetland species contributes to the decline (Stromberg et al. 2007). Extensive piped drainage of catchments via tile drains or open ditches in agricultural lands and storm drains in urban areas may bypass the riparian zone entirely (Figure 9.5). For example, urbanization often results in deepening of the water table in riparian zones, due to diversion of flows (Groffman et al. 2003). Impervious surfaces in the uplands, including pavement, rooftops, and compacted soil amplify peak flows to streams or riparian zones, creating flash floods. High peak flows during storms can cause channel down-cutting and erosion of stream-bank sediments, leading to hydrologic disconnection of the riparian sub-surface from the stream channel (Paul and Meyer 2001).

Hydrological disconnection also occurs due to direct modification of stream channels and riparian zones. Levees built to protect settlements and farms from floodwater may separate a substantial fraction of the riparian area from the action of fluvial processes. This has consequences of eliminating sediment accrual within riparian zones, and reduces flood mitigation and groundwater recharge, because water is flushed more rapidly through the stream channel. Bank stabilization, rip-rapping, and lining of channels have similar consequences and, importantly, result in lowered water tables, restricting water availability in shallow riparian soils (Groffman et al. 2003). Finally, dams alter the hydrologic regime

Figure 9.5:
A buffer strip (grass plus a narrow woodland strip) in Switzerland, near Lausanne. The buffer protects the stream from surface runoff but, unless tile drains are blocked, subsurface runoff will continue to enter the stream unimpeded

of riparian zones by decreasing peak discharge, and significantly extending the inter-flood interval, or time period between floods.

9.4.2. Biogeochemistry

Changes in the hydrological regime alone alter the biogeochemical functions of riparian zones, because of the multiple roles of water in biogeochemical processes. Vegetation subject to drought stress has reduced capacity for uptake of nutrients, and retention and removal of nutrients by soil micro-organisms slows due to water limitation. Rapid runoff or bypassing of the riparian zone during floods decreases water residence time in the riparian zone, and this decreased contact time of solutes and biota restricts the capacity for nutrient retention. Thus, the timing of nutrient delivery to stream-riparian corridors can shift from baseflow to peak flows with increasing hydrologic modification to the catchment (Table 9.2).

Humans directly manipulate the biogeochemical functions of riparian zones through application of fertilisers and pesticides. Although riparian zones may foster high rates of nutrient retention, this capacity for retention can be exceeded when runoff from fertilised fields and residential stock yards results in high loading of nutrients. In addition to increased downstream transport of nutrients, increased nutrient availability in riparian zones can support

Land use	Percentage nitrate exported in baseflow	Percentage nitrate exported in high flow
Agricultural, forested buffer	94	6
Urban	86	14
Mixed (forest, farmland, urban)	78	22
Mixed (forest, farmland)	58	42
Mixed (forest, farmland)	47	53
Forest/residential	21	79
Urban/suburban	10	90
Farmland, tile-drained	3	97

Table 9.2: *The proportion of total nitrate flux exported by baseflow and high flow for a range of streams draining a variety of land uses. Data assembled by Craig et al (2008). Increasing agriculture and urbanisation in catchments results in a shift in the timing of nutrient delivery from baseflow in forested catchments to high-flow events in extensively modified catchments*

growth of invasive plans. Similarly, although riparian zones may promote retention and breakdown of pesticides, this capacity can be overwhelmed by excessive inputs, especially when the spatial extent of riparian zones has been reduced in favour of other land uses. Finally, novel compounds introduced in agricultural and wastewater runoff may cause increased mortality of biota, with potential consequences for riparian food webs. Wastewater from urban areas that is discharged into rivers after treatment may contain high levels of currently unregulated compounds, such as personal care products, caffeine and antibiotics (Chapter 5). These persistent pollutants often have unknown impacts, but are likely to influence riverine and riparian biota for some distance downstream.

9.4.3. Biota

Introduction of invasive species can significantly reduce the portfolio of ecosystem services provided by riparian ecosystems. Non-native plants in particular are often successful invaders of riparian zones, and can affect biotic interactions directly, as well as alter abiotic conditions. For example, the invasive shrub Tamarix thrives in dryland riparian zones of the South West US, especially those subject to flood suppression (Stromberg et al. 2007). Tamarix is associated with drawdown of the water table and increasing groundwater salinity, conditions that are detrimental to native plants. High densities of Tamarix reduce the structural and species diversity of riparian vegetation, degrading habitat quality for some bird species. Non-native plant species that fix nitrogen increase nutrient availability in riparian ecosystems, even at low plant densities, and have consequences for the capacity of riparian zones to perform the service of nutrient retention.

Humans directly alter the biotic composition of riparian zones through vegetation removal, agriculture, and livestock grazing. Riparian zones are cleared of vegetation during forestry, or in preparation for agriculture. Clear-cuts near streams result in significant increases in nutrient loading to streams; increased stream temperatures, which in turn have consequences for stream biota; and decreased inputs of woody debris, which in intact riparian zones contributes structural habitat and organic matter to the stream. In some regions, crops are planted right to the margins of streams, which eliminates riparian habitat entirely. In urban or suburban areas, riparian flora may be intentionally replaced by non-native species (turf grass, non-native trees and shrubs), creating novel communities of plants. Human use of these parklands may be intense. Finally, introduction of livestock grazing to riparian zones has unintended effects of compacting soil, trampling or consumption of vegetation, and destabilization of stream banks; these can often be an important source of sediment input to the channel and require careful management to exclude stock access if in-stream habitats e.g. fish spawning gravels, are to be protected (Figure 9.6).

9.4.4. Interactive effects

By changing individual hydrological, biogeochemical, or biotic attributes of riparian zones, human activities may have consequences for whole riparian ecosystems.

Figure 9.6: *River bank restoration on the Eden River in NW England. The simple expedient of fencing protects the river bank from erosion as livestock no longer have access. It is, however, necessary to provide drinking troughs as part of the scheme*

For example, hydrological disconnection of streams from riparian zones may limit growth of native plant species, which can result in bank destabilization, a decrease in the nutrient-retention capacity of the riparian zone due to decreased plant abundance or resource limitation of micro-organisms, and a change in the quality of food supporting food webs. Such cascading effects are characteristic of all ecosystems, but riparian zones are particularly subject to feedbacks involving disparate spatial locations, owing to connectedness via hydrological flow paths (Burt and Pinay 2005; Chapter 10). As integrators of all activities on the land, streams are sensitive to a host of pressures including impacts from urbanisation, agriculture, deforestation, invasive species, flow regulation, water extractions and mining. The impacts of these individually or in combination typically lead to a decrease in biodiversity because of reduced water quality, biologically unsuitable flow regimes, dispersal barriers, altered inputs of organic matter or sunlight, degraded habitat and so on. Despite the complexity of these interactions, a large number of stream restoration projects focus primarily on physical channel characteristics. Palmer et al (2010) argue that this is not a wise investment if ecological recovery is the goal. Managers should critically diagnose the factors impacting an impaired stream and prioritise those problems most likely to limit restoration (Chapter 11).

9.5. Riparian zone destruction and restoration

In intensively managed areas like city centres and suburbs, streams, rivers and riparian zones may bear little resemblance to their natural character. Small streams are buried, larger ones are channelized and all riparian vegetation may be removed. Extractive activities take place in the floodplain or channel, often removing vast quantities of material as aggregate for construction and leaving great pits that fill with water. Here, the centuries of work of the alluvial system is exploited for useful materials, but the ecosystem has been transformed and a return to its prior state is extremely unlikely, even with intervention. Highly channelized and hardened river banks require continuous vigilance and repair in the face of flooding. On the other hand, recent decades have seen massive efforts at river restoration, many of which provide a cosmetic fix to a degraded system but do not restore underlying ecosystem functions and services (Bernhardt et al. 2005; Palmer et al. 2010; see also Chapter 6). For example, in arid Phoenix, Arizona, USA, riparian restoration projects are *de rigueur*, yet none of these projects relies on restoration of the natural flow regime of the river and all are instead dependent upon imported water to maintain planted riparian vegetation.

We must, however, end on a positive note. Modern legislation to manage river basins, such as the European Water Framework Directive (WFD: 2000/60/EC) tend to adopt a holistic approach focusing on the achievement of "good ecological status". The WFD is formulated to favour functional aquatic habitats as well as potable drinking water. As noted at the start of this chapter, there is an intuitive assumption that the condition of the stream and the condition of the riparian zone are intimately linked. Thus, protection of the riverine environment demands, almost by definition, that full attention is paid to the quality of the riparian zone. Rehabilitation of natural habitats, restoring wetlands and removing inappropriate land uses in the riparian zone can all contribute to a sustainable future for our rivers and their habitats. In the decades to come, climate change may become the main driver of long-term change in river ecology but in the short term, land use seems to be a more important factor. Restoration of riparian zones to their natural condition is a great challenge to scientists, regulators, politicians and land owners alike but may nevertheless provide the most cost-effective means of managing our river basins going forward. Probably, a traditional approach to nature conservation in riparian zones based on biodiversity and naturalness is insufficient in itself, but a wider perspective, considering all the benefits to the river system, provides justification for maintenance of riparian zones in good ecological status.

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Chapter 10

Ecological Connectivity for River Conservation

Deb Finn and Jeremy Monroe

Connectivity in river ecosystems can refer either to organisms and nonliving materials moving within and among river networks *(network connectivity)* or to nutrients and energy moving through food webs and linking aquatic with terrestrial or marine ecosystems *(web connectivity).* By nature, rivers are complex *networks of webs* in which multiple dimensions of connectivity interact. Many human endeavors disrupt these networks of webs, but thoughtful conservation management can help maintain sustainable levels of connectivity.

10.1. Fish and amphibians in the Necklace Lakes

The Necklace Lakes of Montana, USA are threaded like a string of pearls along a chain of small streams in a broad wilderness basin. In the lakes along this chain, trout have thrived for thousands of years (native cutthroat trout for most of that time, and introduced species like rainbow and brook trout more recently). Also present are several amphibians, including the long-toed salamander and various frogs. Interestingly, the necklace appears broken in some places, as a number of pearls strewn nearby are separated from the chain. These pearls are lakes without small stream outlets or inlets. A biologist out mucking around the Necklace Lakes basin will notice right away that, depending on the time of year, various stages of amphibians from egg to adult thrive in much larger densities in the separated lakes than in those occupying the intact necklace chain.

A lack of *connectivity* to the main necklace chain renders these separated lakes fishless. Trout require aquatic habitat at all life stages, and because they cannot survive on land individual fish cannot make the move over even short distances to colonize neighboring separated lakes. Conversely, trout occupy all lakes along the intact necklace. The historically fishless state of the separated lakes has allowed amphibians to thrive in a state of release from both predation and indirect negative effects of fish. Adult amphibians, however, can move across land, so the concept of connectivity for these animals is not the same as that for fish. Hence, amphibians can be found in the sub-par habitat along the necklace chain, but it is likely that thriving populations in the fishless habitat of the separated lakes supplement the necklace populations regularly. At the landscape scale, fish and amphibians enjoy a stable coexistence in the Necklace Lakes basin, thanks in large part to contrasting definitions of connectivity for these two groups.

10.2. What is *connectivity*?

Network connectivity can be described both within and among independent river networks, encapsulates three spatial dimensions within networks, and typically increases with temporal flow pulses

If connectivity has different meanings for amphibians and fish, is there a general definition for the word? Typing *connectivity* into an internet search engine will give an idea of how the word is used most commonly – and what else do we find these days but references to computers and the internet (see Box 10.1). Ecologically speaking, connectivity has an analogous interpretation in terms of movement of cohesive *packets* from one place to another – only in the ecological realm, these packets are either organisms moving across a landscape (as the Necklace Lakes fish and amphibians) or materials of biological importance (e.g. nutrients and energy-containing molecules) moving either through a landscape or from one organism to another through a food web. Landscape connectivity is intuitive when we humans can see a physical pattern that might directly translate, for example, to an organism's movement ability. Intuitive examples include the connected vs. separated lakes in the Necklace Lakes basin, large bridges over or passages under major roadways to allow movement of wildlife, or the lack of connectivity between oceanic islands or between stream segments upstream vs. downstream of a large waterfall. Colonization of the New World via the Bering Land Bridge between northeastern Asia and northwestern North America provides an intuitive human example. The land bridge was exposed during low sea levels of the last Ice Age, increasing connectivity for terrestrial organisms and allowing movement of human populations from what is now Asia into previously uninhabited continents.

Internet connectivity facilitates social connectivity

Box 10.1

There is a strong parallel between current widespread definitions of connectivity and the definition of "IP" (Internet Protocol), which refers to the transfer of packets of electronic information between two endpoints. Social networking sites like Facebook tout the benefits of facilitated connectivity of people around the world via their internet-based service. Indeed, 845 million people (as of Feb 2012) enjoy the ease of globally communicating anything from their breakfast menu and baby photos to ideas seeding revolutionary uprise thanks to the increased international social connectivity that Facebook facilitates.

But landscape connectivity is not always intuitive. Sometimes, for example, connectivity for one organism relies on the presence and activity of another organism, as is the case for many freshwater mussels (Chapter 6). Adult mussels are sedentary filter-feeders but their larvae are capable of movement away from the natal site by temporarily parasitizing a fish's fins or gills. This arrangement is typically species-specific, so the presence and movement behavior of particular fish species dictate how far mussels are able to move across the landscape (or "riverscape", "riverine landscape"). Two riverscapes might look equally connected to the human eye, but one might have high connectivity for a mussel species owing to an abundance of its host fish species, and the other might have lost the host fish species resulting in extremely low connectivity for the mussel. So it is important to remember that connectivity is not solely a property of a landscape.

Rather, it is a property of the interaction of a landscape and an organism's movement-related traits (Taylor et al. 1993; Ricketts 2001).

Connectivity has a slightly different interpretation in the context of food webs, which are descriptions of which organisms eat which in an ecosystem. Food web connectivity should be equally intuitive, however: organisms still play the key roles, but it is their trophic interactions (who eats whom) that dictate connectivity of nutrients and energy (i.e. food) through a food web and, potentially, across ecosystem boundaries. Food web connectivity (as we apply the term in this chapter) increases when two or more ecosystems that are traditionally considered separately (e.g. aquatic and terrestrial) are linked via cross-system flows of energy and nutrients (Polis et al. 1997). Globally, about 1 billion people living in coastal communities depend on ocean fish and shellfish as a primary food source. Hence these terrestrial humans, via regular consumption of ocean-produced energy and nutrients, rely on original sources of primary food production in the ocean. In this example, ocean ecosystems are said to *subsidize* terrestrial ecosystems via the high degree of food web connectivity achieved by human fishing and eating behavior. In the Necklace Lakes basin, an adult frog may forage terrestrially and consume many flies that themselves consumed primarily the food produced by terrestrial plants. The frog's foraging success then allows her to lay 1,000 eggs, most of which get consumed within days by an introduced rainbow trout living in the lake along the edge of which the frog laid her eggs. Here, terrestrial production has subsidized the aquatic food web via trophic interactions among forest plants, flies, frogs, and fish.

10.3. Describing connectivity in river ecosystems

Rivers have a distinctive physical structure that has served to increase connectivity among human populations since pre-history. A glance at a regional map provides insight: a river's branching-linear appearance could lead one to mistake it for a series of roads (and the purpose of roads is to increase connectivity among human populations). The map also reveals that most cities lie on rivers, and rivers (like roads) typically link non-coastal cities (see Figure 10.1). This arrangement is of course no accident. Rivers not only provide essential consumable resources (food and water); they also greatly enhance trade and transport among human settlements.

The unique structural organization of river ecosystems into hierarchically branching networks is fundamentally the same worldwide wherever rivers occur, from rainforests to deserts and even in urbanized areas. The structure of smaller branches merging in pairs to form larger branches initiates with the tiniest upland streams and continues incrementally until a large river reaches an outlet. Hence, there is a physical continuity between a very large river and the multitude of smallest streams ("headwaters") spidering across the uplands of the catchment feeding it. Conversely, small headwaters might be very near to one another on a landscape but not be connected hydrologically because they occur on opposite sides of a drainage divide (and therefore occupy different catchments). This unique structure of river ecosystems across landscapes has strong implications for connectivity via movement of organisms. Those organisms unable to leave the aquatic habitat (e.g. most fish) have no connectivity across catchment boundaries, while those organisms capable of terrestrial movement (e.g. amphibians) do not recognize such strict limitations on connectivity.

The branching network structure of river ecosystems is reminiscent of various biological transport systems within individual organisms (Lowe and Likens 2005). A common structural analogy is a tree, in which millions of tiny veins within thousands of leaves each are directly connected to a single large trunk. The circulatory and respiratory systems of humans are similarly arranged, with millions of tiny capillaries connected directly to one of the two largest veins feeding the heart (circulatory system) or millions of tiny alveoli within the lungs directly

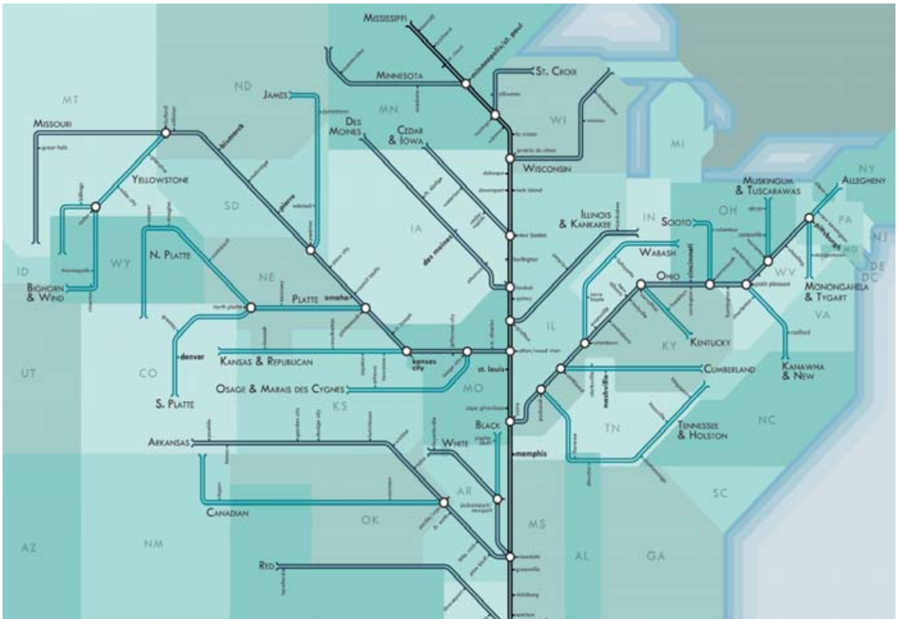


Figure 10.1:
One of mapmaker Daniel Huffman's "Rivermaps", depicting the major drainages of the Mississippi River network. Huffman represents river networks as transportation corridors connecting cities, applying the style developed by Harry Beck in the 1930s for use in public transport maps (think London Underground). The style intentionally distorts the "true geography" to simplify and clarify connectivity

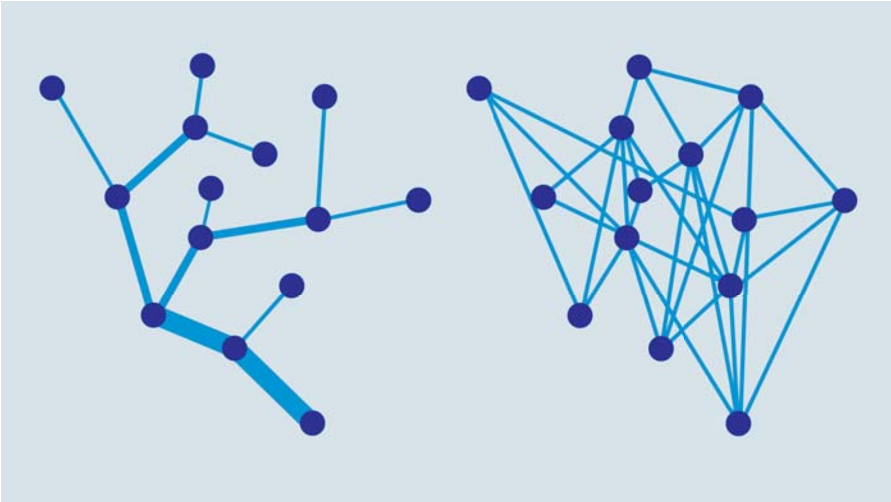


Figure 10.2: *Conceptual diagrams to differentiate the terms "network" (left panel) and "web" (right panel), as they are used in this chapter. Each panel contains the same number and orientation of nodes (dark blue circles). Note two key differences: 1) Connectors linking nodes in a network are restricted to two upstream and one downstream, but the number of connectors linking nodes in a web are limited only by the abundance of other nodes. 2) For our purposes, sizes of connectors in a network are determined by their relative location, where those farthest upstream (e.g. headwaters in a river network) are smallest and have closest interaction with the surroundings. Sizes of connectors in a web follow no such restrictions*

linked to the single, large trachea (respiratory system). In addition to the structural similarities among these network-like systems, there are functional analogies. Namely, in each of these examples, the most intimate interactions between the network and the surrounding environment occur within the smallest branches. Tiny veins in tree leaves drop off water molecules and pick up newly produced sugars from photosynthesis in the surrounding leaves. Tiny capillaries in the human circulatory system are the exchange sites for oxygen, carbon dioxide, and other nutrients and waste products with bodily organs and tissues. And the alveoli of the lungs are the sites of gas exchange (oxygen for carbon dioxide) with the blood. The small headwaters of river ecosystems also have a particularly intimate connection with the terrestrial landscape in which they are embedded, and strong terrestrial/aquatic interactions occur at these locations. Hence: connectivity between aquatic and terrestrial ecosystems is amplified in unimpacted headwaters.

For the remainder of this chapter, we will refer to the movement of organisms and nonliving materials of biological importance within and among river ecosystems as *network connectivity,* after the unique structural template of the river itself as a branching network strongly influencing the movement of organisms and materials. Food web connectivity in rivers we will shorten simply to *"web connectivity",* to contrast structurally with network connectivity. A web structure, as in a food web, does not have the same branching, hierarchical restrictions as a network (Figure 10.2). One node in a food web, an invertebrate consumer in a small stream for example, can be connected to a multitude of other nodes in the web (e.g. it might be a generalist consumer connected to leaves, algae, and moss, and it might be connected as prey to several large insects, a crayfish, three species of fish, and a few birds and bats). One node in a stream network, however, has a maximum of three connections: one downstream and two smaller upstream branches. We will see that both network and web connectivity are important to consider for river conservation.

10.4. Network connectivity

We must, in fact, not divorce the stream from its valley in our thoughts at any time. If we do, we lose touch with reality.

H.B. Noel Hynes, 1975

10.4.1. Models of organism movement within and among river networks

Recall that connectivity associated with organism movement is an interactive property of landscape structure and the movement-related traits of the organism in question. The Necklace Lakes example revealed different degrees of connectivity for amphibians *vs.* fish occupying the same structured landscape, because these two groups have different capacities for movement among lakes. When we move to consider connectivity across a much larger spatial extent, potentially including multiple independent river networks and a stream-size range from small upland headwaters to large outlet rivers, another factor to consider is the degree of habitat specialization of the focal organism.

Habitat specialization essentially describes the degree to which a species is restricted to a particular habitat type or, in rivers, a particular zone of a network. Suffice here to differentiate two major groups: *specialists* and *generalists*. The small headwaters at the myriad upper tips of networks harbor a great number of habitat specialists within river ecosystems. That is: many species are found only in headwater habitat. There could be multiple explanations for this pattern. One may be cold stenothermy (i.e. narrow temperature requirements on the low end of the thermometer). More generally: headwaters, tightly linked to the terrestrial ecology of the small basins they drain, provide unique habitat conditions that are also highly independent of one another, creating a mosaic of local habitat types even within a single river network. Examples include but are not limited to varying geological setting (e.g. granite *vs.* limestone), different water sources (e.g. groundwater *vs.* snowmelt), or different riparian conditions (forested *vs.* unforested) across headwaters in the same network. Streams occupying lower positions in the network blend the varied characteristics of the multitude of smaller upstream segments and hence typically do not exhibit the degree of habitat uniqueness found in headwaters. It follows that a majority of habitat specialists in rivers are thought to occur in headwaters, and many of the species found in larger streams are more often habitat generalists.

Four models have been developed to describe connectivity within and among river networks according to an understanding of movement traits and habitat specialization of aquatic organisms (Figure 10.3) (Hughes et al. 2009; Meffe and Vrijenhoek 1988). Typically, a researcher makes a hypothesis about which of the four models might apply to a particular species, and the most common way to test the hypothesis is by collecting genetic samples from many individuals of the species, in several locations across multiple river networks. The genetic data represent some highly variable marker (or markers) in the genome, such that differences among individuals of the species are readily detectable. Sec-

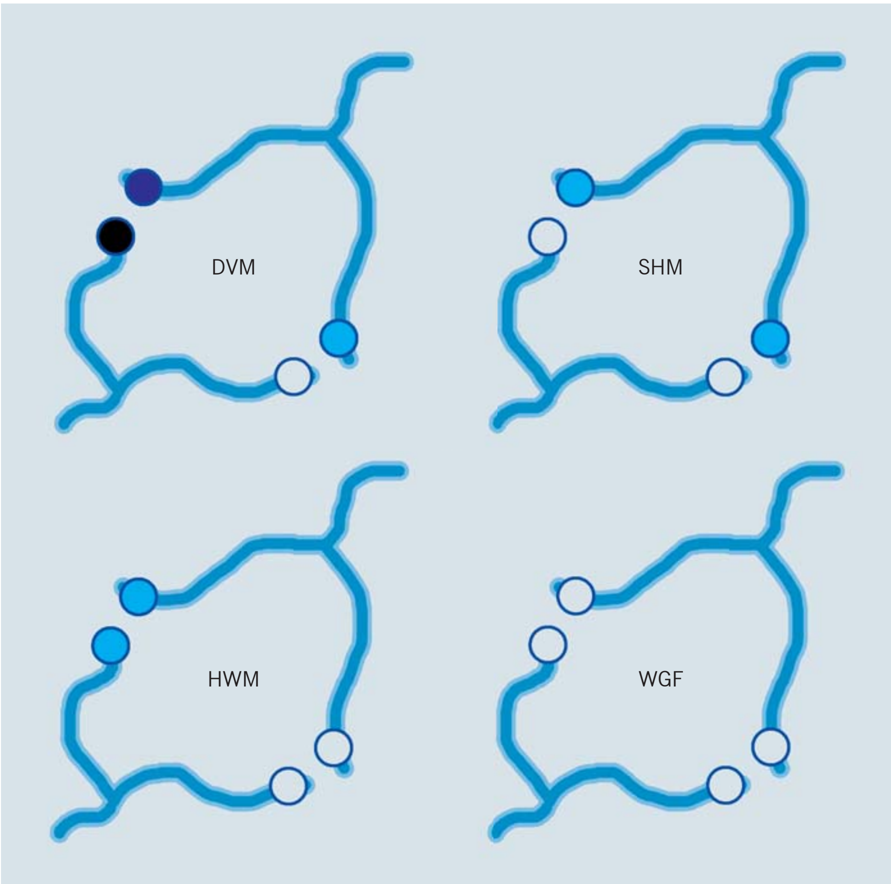


Figure 10.3: *Conceptual diagrams of four different connectivity models, each applied to two simple networks that flow in opposite directions from headwaters originating in close proximity (e.g. same mountain range). Circles indicate locations of stream-dwelling animal populations; colors indicate similarity via presumed gene flow, such that populations of the same color experience the maximum* connectivity. *In order from left to right: Death Valley Model (DVM), stream hierarchy model (SHM), headwater model (HWM), widespread gene flow (WGF). See text for examples of each*

tions of mitochondrial DNA have been probably the most commonly applied genetic markers to date for testing hypotheses about connectivity in rivers. Tests assess statistically how genetic differences are distributed across space, under the assumption that increased connectivity leads to increased genetic similarity among sampled locations – and vice-versa. *Gene flow* is a term analogous to connectivity indicating statistical evidence that genes are moving regularly from one location to another. Gene flow is a product of individuals physically moving and then successfully reproducing in the new location.

The first connectivity model, the *stream hierarchy model* (SHM) describes high connectivity internally within river networks from headwaters to large rivers and low connectivity from one network to the next. The SHM is the most intuitive of the movement models in rivers because the stream network itself is the major movement corridor, and organisms are presumed to be habitat generalists within the network. Hence, for species that follow the SHM, connectivity is higher within than among networks. The "hierarchy" in the SHM refers to the hierarchical structure the model can take, analogous to the hierarchical structure of stream networks: subnetworks within larger networks have increased internal connectivity. Animal species that typically follow the SHM are those having little or no ability to leave the aquatic environment – but also those that are not strict habitat specialists (e.g. in headwaters). These include many species of fish, as well as many invertebrates that have little possibility for terrestrial movement (e.g. mussels, aquatic insects lacking a terrestrial adult).

The second model is termed the *headwater model* (HWM). Fundamentally the inverse of the SHM, the HWM is expected for stream-dwelling species that are habitat specialists in headwaters and have some capacity for overland movement, typically by crawling or limited flying – e.g. amphibians, many crayfish, some aquatic insects. The HWM predicts that connectivity will be strongest among groups of nearby headwaters; i.e. those that are "crawling distance" apart, regardless of hydrologic connectivity. Such spatial clumping of headwater streams typically occurs in topographically complex landscapes, particularly when multiple island-like mountains, mountain ranges, or other uplifts in a region are separated by lower-elevation "seas" representing a different habitat type. The Madrean Sky Islands are a series of small mountain ranges rising above a sea of low desert in southern Arizona, USA and northern Sonora, Mexico that provide a compelling case of the HWM for both a stream insect predator, the giant water bug *Abedus herberti* (Finn et al. 2007), and the canyon treefrog *Hyla arenicolor* (Barber 1999). Headwaters and larger, lowland streams, exhibit strong habitat disparity in this region, as many headwaters have permanent surface water, and the lowland desert streams are intermittent. Although a group of headwaters originating on a single mountain range could occupy multiple independent river networks (and headwaters from multiple mountains can occupy the same network), connectivity for both bug and frog is much stronger among headwaters sharing a mountain than among those sharing a network. To put this pattern in perspective: *A. herberti* mountain-range populations differ genetically from one another by up to 2% across this small region. The average genetic difference of two humans, randomly selected from the entire world, is approximately 0.4% (both estimated from mitochondrial DNA). Clearly, connectivity is quite low for *A. herberti* among mountain ranges. There are several other headwater specialists in the Madrean Sky Islands for which connectivity has yet to be assessed, but it is likely that the HWM holds for many of these as well.

The final two models represent opposite endpoints, between extremely low (the *Death Valley Model*, DVM) and extremely high connectivity *(widespread gene flow,* WGF). The DVM is an appropriate metaphor that implies aquatic habitats that are completely isolated from one another, no matter the landscape or river-network structure (e.g. headwater springs in Death Valley, USA). The DVM essentially represents an extreme of the HWM, suitable for cases when headwaters are isolated from one another to the extent that only very rare or zero among-site movement is possible. This situation could arise in one of two ways. First, the species in question, e.g. fish, have no ability to move from one aquatic habitat to another. This is the situation for the DVM's namesake, small fish occupying Death Valley springs with no surface water connection. Second, the focal species has a limited capacity to move among habitats, but the landscape surrounding the aquatic habitat is either too extensive or too inhospitable to allow successful overland movement. An example here is a rare, non-biting black fly *(Metacnephia coloradensis)* occupying the lake outlet streams of only very large, high-altitude lakes in the Rocky Mountains – the adult stage has limited capacity to fly, required habitats are rare, and the landscape separating them is treacherous, so connectivity among the few populations of this species is thought to be effectively zero.

Widespread gene flow occupies the opposite end of the connectivity spectrum and is expected for species having either a highly mobile terrestrial stage or traits allowing passive dispersal by either wind or temporary association with mobile animals such as water birds (Figuerola and Green 2002; Maguire 1963). Charles Darwin performed classic early studies demonstrating both the diversity of plant seeds embedded in the mud on a duck's legs and the association of some otherwise sedentary invertebrates (even as large as snails) with the legs of water birds. Such examples are more common than one might expect, and they often account for observed patterns of widespread gene flow in aquatic organisms that lack the ability to disperse among catchments under their own power. Conversely, many caddisflies (Trichoptera) are strong fliers that can disperse long distances by themselves. An example is *Plectrocnemia conspersa*, a common caddisfly occupying upland streams in northern Europe (Wilcock et al. 2001). Although this species is a habitat specialist in small headwaters, connectivity is strong across most of its extensive range. As we hinted earlier, however, the structure of the intervening terrestrial landscape can strongly influence connectivity, even for flying aquatic insects. An eastern North American mayfly *(Ephemerella invaria)* is not a particularly picky habitat specialist, and given a forested terrestrial landscape (which provides an ideal environment for overland flight of many insects), widespread gene flow is expected. However, deforestation to the extent of leaving intact forest only along riparian buffer zones has reduced connectivity in recent years to the stream corridors, effectively changing the connectivity model for *E. invaria* to some combination of SHM + HWM (Alexander et al. 2011). Changing land use within drainage basins therefore is an important consideration regarding network connectivity in river ecosystems.

10.4.2. Temporal "pulses" and three spatial dimensions of connectivity within networks

Naturally, river habitat is defined by the spatial distribution of flowing water, the essential "ingredient" in river ecosystems. The presence of flowing water allows us to delineate where riverine and terrestrial habitats begin and end, and it strongly influences connectivity of both organisms and biologically important nonliving materials. But what if a river has no surface water? Or if flow has ceased, resulting in only a few separated pools of standing water along the streambed? Intermittent streams and rivers contain flowing surface water only during certain parts of the year, and some ephemeral streams flow only when it rains. A key to understanding such systems is to appreciate that nearly all river ecosystems undergo natural flow "pulses" similar to the pulsing of blood through the circulatory system (Junk et al. 1989; Poff et al. 1997).

The "heart" controlling pulses in rivers is the annual cycle of precipitation and/or snow- or ice-melt characteristic of a region. Rivers and streams – whether perennial, intermittent or ephemeral – expand and contract in size with this temporal cycle (Chapter 2). Hence, streams that have temporarily lost surface flow (but nonetheless nearly always contain groundwater not far below the surface) are simply in-between pulses, when aquatic habitat is contracted to a minimum volume. The pulse in rivers (compared to our circulatory system) may or may not be highly predictable through time. Rivers fed by mountain snowmelt, for example, have predictable pulses in late spring when rapid melting occurs. Conversely, some desert and prairie streams are considered "flashy", flooding unpredictably with chance rainstorms. Either way, pulses not only increase the volume of liquid water, they also increase connectivity in three key spatial dimensions of river ecosystems: *longitudinal* (i.e. upstream-downstream), *vertical* (between groundwater and surface water), and *lateral* (between riparian/floodplain habitat and the main stream) (Figure 10.4) (Ward 1989). In the prior section, we focused on connectivity via organism movement patterns across large spatial extents, including multiple river networks and the landscapes that contain them. Here, we emphasize single networks and the importance of flow pulses to connectivity in the longitudinal, vertical, and lateral dimensions.

Longitudinal connectivity. When a pulse returns surface flow to an intermittent stream, hydrologic connectivity between headwaters and larger rivers is reestablished. Some stream-dwelling animals with a terrestrial stage (e.g. many insects) will disperse longitudinally along streams lacking permanent surface flow, and some very small organisms and dissolved materials can move longitudinally through groundwater under the streambed – but a surface water connection substantially amplifies longitudinal connectivity and is clearly necessary for movement of larger aquatic organisms (like fish) and nonliving materials (like leaves and other organic debris). The stream hierarchy model (above) also assumes the potential for relatively unrestricted movement through surface waters in both upstream-to-downstream and downstream-to-upstream directions. There is substantial evidence that several fish species move upstream to spawn in intermittent headwaters, where the porous cobble streambeds that provide ideal substrate for constructing nests ironically also are more likely to lose surface flow during dry periods. Coho salmon in the US Pacific Northwest often spawn in such streams. Juveniles remain and rear in isolated standing pools during periods when surface flow disappears, then they follow the network

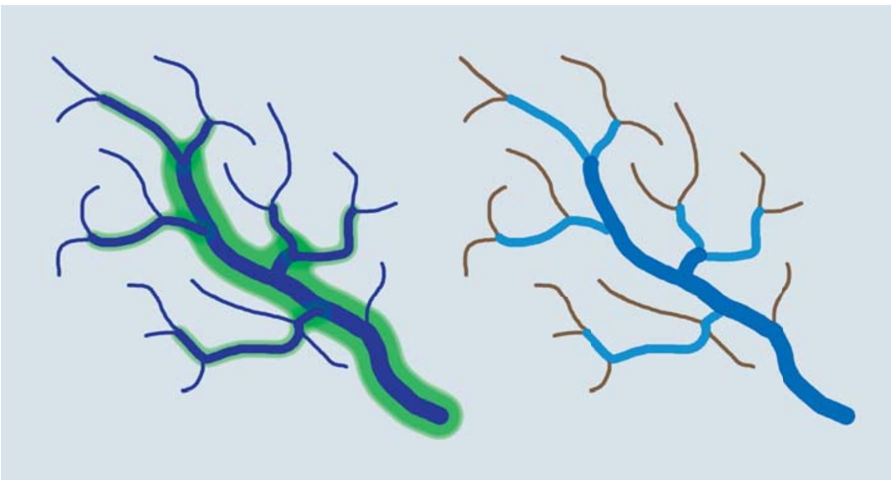


Figure 10.4: *Two extremes of the flow pulse in a conceptual river network: a pulse of high flow (flood) on the left, and low-flow (drought) conditions in-between pulses on the right. Blue/brown shading in channel indicates degree of surface flow (maximum = darker blue; minimum = brown, no surface flow), and green indicates extent of lateral and vertical connectivity via inundation of floodplains and hyporheic zone*

downstream to the sea during a subsequent flow pulse (Wigington et al. 2006). Coho, an anadromous species that spends most of its adult life in the ocean, illustrate clearly the importance of longitudinal connectivity in river networks. All salmon rely on connected waterways to move between the ocean and spawning habitats in small streams.

Longitudinal connectivity is also important with regard to transport of food materials (i.e. energy in the form of suspended and dissolved organic material, and nutrients) downstream from headwaters to larger rivers. Headwaters, acting as the capillaries of river networks, readily interact with the terrestrial environment in which they are embedded, and these interactions can have far-reaching effects at significant distances downstream. In many headwaters, more organic material enters the stream from the surrounding catchment than is consumed. Conversely, in larger rivers little organic material enters directly from the terrestrial environment, but there are typically more consumers. Hence, organic material transported longitudinally in various forms can supply essential energy sources to downstream food webs. Headwaters also appear to play a significant role in moderating water quality throughout river networks (Naiman et al. 1987). A study of a number of prairie streams of different sizes in Kansas, USA revealed that the best predictor of water quality (particularly nutrient load in this highly agricultural region) was riparian land cover associated with the smallest headwaters – no matter their longitudinal distance upstream (Dodds and Oakes 2007). That is: the condition of the riparian zone adjacent to headwater streams has far-reaching downstream effects on water quality.

Vertical connectivity. Groundwater is often overlooked as part of freshwater ecosystems for the simple reason that we can't see it. However, the *hyporheic* zone (literally "below the flow", but often extending laterally some distance away from the stream channel) of streams and rivers is essential both as habitat for organisms and as a location for processing nutrients and organic material exchanged with the surface-water environment. Hence, vertical connectivity plays a key role in river ecosystems. Aquatic animals that are small or resourceful enough to travel the "interstitial highway" of contiguous aquatic habitat surrounding cobbles, gravel, and even sand of the hyporheic zone do so for different reasons. Some animals that are typically members of the surface-water community may use the hyporheic zone as a temporary refuge, either during particularly strong flow pulses (floods) or during droughts in intermittent streams. In streams with predictable pulse timing, the development rates of some invertebrates are timed such that they are still at a small enough stage of development to occupy the groundwater habitat when it is useful as a refuge. Other hyporheic occupants may specialize on this habitat and spend all or the majority of their life cycles there. Crustacean *meiofauna* (loosely defined: larger than microscopic but small enough to pass through a 1mm mesh), which lack a terrestrial phase, are common examples (e.g. amphipods, copepods). However, there are also remarkable examples of long-term hyporheic macrofauna, many of which are burrowers that make a living in sandy hyporheos. The caddisfly *Pedomoecus sierra* achieved some notoriety at desert springs of the Great Basin (US) because researchers commonly collected the terrestrial adults and yet could rarely find the larvae in collections from the springs themselves (Myers 2011). Finally the researchers discovered that *P. sierra*'s entire larval (and pupal) life is spent burrowed within hyporheic sand. It appears to specialize on eating microbial growth attached to the sand grains. Stiff hairs and spines on the larvae both assist burrowing and prevent sand grains from entering the rock case.

Microbial growth is ubiquitous in hyporheic and groundwater habitats. Microbes can thrive in the interstices and in the absence of sunlight given a reliable source of dissolved organic material, which typically is supplied by the surface water. Nutrients, conversely, tend to be more concentrated in groundwater than surface water, and in locations where "upwelling" (net flow from groundwater to stream channel) occurs, streambed algae often grow rather densely (Boulton et al. 1998). However, nutrient dynamics at the surface/groundwater boundary are complex and typically quite situation-specific. Colonies of nesting birds occupying small catchments, for example, can drastically inflate nutrient concentrations in the groundwater, and, ultimately, the stream. Alternatively, high densities of spawning salmon can result in nitrogen flow from streams, where the fish spawn and eventually die, to the connected groundwater. Nutrient flow in this direction can supplement primary production, including tree growth, in the adjacent riparian zone (Chapter 4).

Lateral connectivity. During a flow pulse, the increased volume of water often exceeds the bounds of the river channel and inundates riparian habitat. The extent of this lateral inundation of what is known as the *floodplain* depends on the magnitude of the pulse and the degree to which the river is constrained (e.g. canyon sections of rivers have little leeway for lateral expansion). Inundated floodplains in their natural state can be quite extensive and complex, typically comprising a mosaic of surface-water habitats, from small standing pools to large braided channels that only flow during floods. This complexity combined with the lateral connectivity achieved between main channel and floodplain during flow pulses greatly enhances biological diversity and productivity of river ecosystems.

Major biological implications of lateral connectivity vary according to timing with respect to the flow pulse (Junk et al. 1989). Much directed movement from river channel to floodplain occurs on the approach to and during the peak stages of the pulse, when river-borne nutrients get deposited on the floodplain, and many stream-dwelling animals move laterally to use the more benign aquatic conditions of the floodplain as a refuge from high flows. A variety of insects adapted to conditions in regularly flooding streams actually extend the concept of lateral connectivity beyond the floodplain and well into the terrestrial landscape when they use heavy rainfall as a cue to crawl away from the stream and into the uplands in anticipation of the flood pulse (Lytle and White 2007). Closely following the peak of the pulse, newly deposited nutrients stimulate production in wetted floodplain habitats, terrestrially derived organic materials on the floodplain become available as food resources to the aquatic ecosystem, and many animals remain to take advantage of the rich environment and use the aquatic habitat for spawning and rearing of young. The intense biological activity in the floodplain generates organic material and nutrients that can then be transferred laterally to the main river channel. In rivers with intact floodplains, lateral supplements of organic material to the stream channel following a flow pulse can substantially exceed longitudinal supplements from the headwaters. Like headwaters, the mosaic of smaller aquatic habitats on the floodplain interacts closely with the surrounding terrestrial environment, essentially playing the same "capillary"-like role as headwaters but in a location directly connected to potentially very large rivers. Lateral connectivity can therefore have a strong influence on river ecosystem functioning .

10.5. Web connectivity

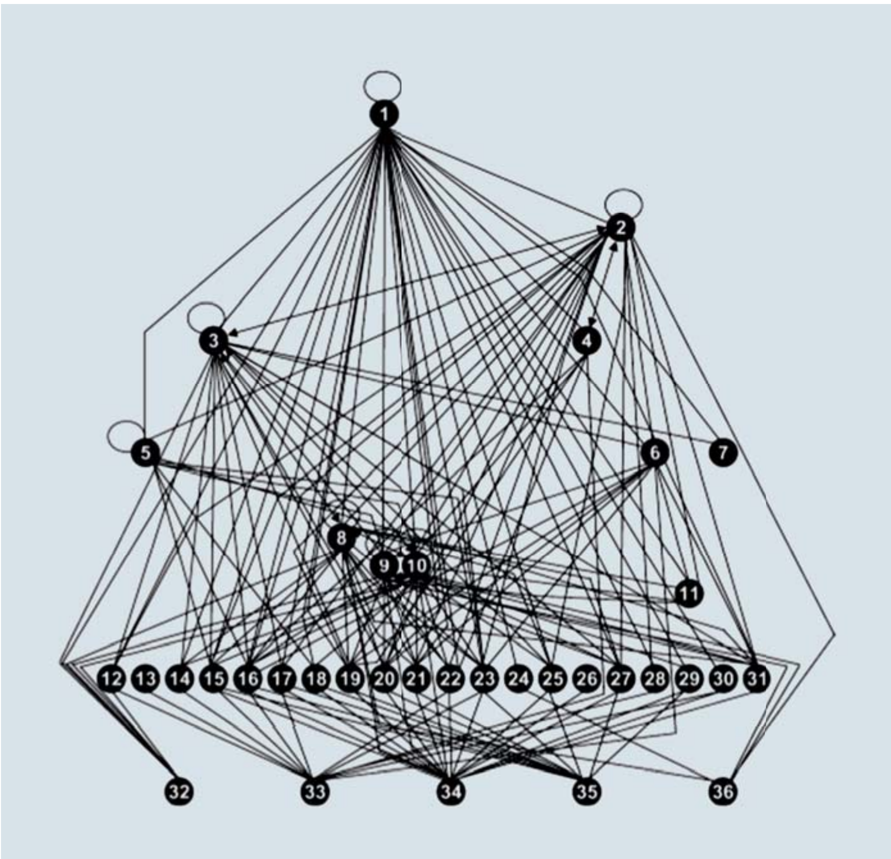
> *Food is the continuum in the song of the [Río] Gavilán. I mean, of course, not only your food, but food for the oak which feeds the buck who feeds the cougar who dies under an oak and goes back into acorns for his erstwhile prey. This is one of many food cycles starting from and returning to oaks, for the oak also feeds the jay who feeds the goshawk who named your river, the bear whose grease made your gravy, the quail who taught you a lesson in botany, and the turkey who daily gives you the slip. And the common end of all is to help the headwater trickles of the Gavilán split one more grain of soil off the broad hulk of the Sierra Madre to make another oak.*
>
> Aldo Leopold, 1949
> "Song of Gavilan", in Part II of *A Sand County Almanac*

10.5.1. River food webs

The nutrients and organic material that move among river networks and within their three spatial dimensions form the basis of river food webs. Primary producers such as plants and algae take up essential nutrients like nitrogen and phosphorous in dissolved, molecular forms and "fix" inorganic carbon sources (primarily carbon dioxide from the atmosphere or dissolved in the water) into energy-containing molecules. The combination of nutrients and energy-rich carbon-based molecules in aquatic primary producers and dead organic material then provide the basal food resources for all consumers in river food webs. When a consumer feeds on one of these basal resources or on another consumer (predation), nutrients and carbon effectively move from one node of the web to another. Even food webs in the smallest streams can be quite complex, with pathways connecting basal food resources to multiple levels of consumers and predators in myriad potential configurations (Figure 10.5).

Researchers can trace the connectivity of nutrients and carbon between nodes in a food web with a number of different approaches. Most simply, one can observe the eating habits of animal consumers. This approach is most feasible for larger river-dwelling animals; particularly fish. Fish biologists often don a mask and snorkel to observe eating behavior, and fly-fishing can be a never-ending

Figure 10.5: *A diagram of the food web in Broadstone Stream, a small chalk stream in England. Each black circle is a species or group of similar species; vertical position of black circles indicates trophic position, with primary producers at the bottom, primary consumers second, and so on until the "top" predator; connector lines connect "who eats whom"*



experiment testing which prey items different fish species are choosing to eat at particular times of the year. Another direct approach to evaluate diet is by looking at what consumers have already eaten (i.e. what is in the gut). For smaller consumers, like invertebrates, this typically involves completely removing and opening the gut cavity. For larger consumers, gut contents can be evaluated non-lethally, often by the process of "gastric lavage" (literally "stomach washing"), which forces water into and then out of the stomach to flush out its contents. However, because many of the nodes of a stream food web represent very small-bodied animals, and in some cases it is impossible to determine food items (e.g. the "true bugs", Hemiptera, liquefy prey prior to sucking it through a straw-like mouth appendage), supplementary approaches are often necessary.

One common indirect approach to evaluating who is eating whom in a stream is by referring to a published list of known "functional feeding groups" (FFGs, e.g. Merritt and Cummins 1996) for stream-dwelling invertebrates. These FFG lists allow a researcher to assign to a consumer species the most probable of the important food resources available (e.g. algae growing on the streambed; large organic material, like leaves, from the terrestrial environment; fine bits of organic material suspended in the water column; living animal prey). Typically FFG lists are based on previous research, but they might also be inferred from other aspects of the species' biology, such as the structure of its mouthparts, its behavior, or how closely related it is evolutionarily to another species for which the FFG is better understood. FFG lists are helpful in determining river food web structure, but a key drawback is that many aquatic invertebrates have a more generalist (i.e., omnivorous) diet than we often would like to admit. For example, it is clear that any animal that makes a living filtering small particles from the flow with either a constructed silk net (some caddisflies) or specialized appendages (black flies, many others) is eating fine bits of organic material. But this filter-feeder also could be undiscerning to the degree that it will eat small animals that have become detached from the streambed and drift into the filter apparatus, as is the case with some black fly larvae that have been observed to eat small, drifting midges. This example is one of many that reveals the truly omnivorous feeding nature of many stream-dwelling invertebrates.

Another indirect option for tracing the pathways of nutrients and carbon though river food webs is by evaluating chemical aspects of the elements themselves. An element's isotopes vary in size (mass) by a minute degree that can be detected with an instrument called a mass spectrometer. Of particular relevance to river food webs are isotopes of nitrogen (N) and carbon (C). Proportions of heavier to lighter N isotopes in an organism provide a measure of how predatory that organism is. With each "step" in a food chain, from primary producer

Web connectivity links river ecosystems to other major ecosystem types (terrestrial, marine) via cross-boundary transfers of nutrients and energy through food webs

to consumer to potentially multiple predator levels, the ratio of heavy to light N isotopes increases at a predictable rate. Hence, ratios of N isotopes could allow us to separate filter-feeders, as above, that often consume animal prey from those that only consume fine organic material. Proportions of heavier to lighter C in an organism can give an idea of whether its basal food resources were mainly of terrestrial or aquatic origin, as primary production in these two environments results in different ratios of C isotopes. Evaluating C and N isotope ratios in concert therefore provides an opportunity to disentangle to some extent the diets of generalist consumers in river food webs. Carbon isotopes also can help evaluate web connectivity between terrestrial and aquatic food webs; i.e. to what extent one ecosystem subsidizes the other.

10.5.2. Aquatic/terrestrial web connectivity

Ecologists have long recognized the importance of terrestrial subsidies to river food webs, and headwater streams embedded in forested basins are the prime archetype (Wallace et al. 1999). Especially during autumn leaf-fall, such streams become choked with leaves and other organic material, a massive food supply for primary consumers living in these streams and, sequentially, their predators. Many smaller bits of organic material move longitudinally from headwaters with downstream flow and can supplement food webs of larger streams that may have less local input of organic material (Vannote et al. 1980). Connectivity to floodplains along larger rivers of the network also can supply substantial amounts of terrestrially derived organic material to fuel aquatic food webs.

Although forested headwaters are the model of heavy terrestrial subsidization of river food webs, researchers now recognize a strong signal of terrestrial primary production even in many treeless headwaters (e.g. in arctic or alpine tundra). This terrestrial signal (determined by carbon isotope ratios in consumers) increases further in streams with proportionally more wetlands in their catchments. Wetlands contribute substantial dissolved organic material (primarily of terrestrial origin) to streams, so it is likely that these stream food webs are heavily reliant on microscopic fungi and bacteria (together: microbes) that decompose dissolved organic material, using it as a source of food energy (Dekar et al. 2012). With increasing rates of decomposition, microbes increase in biomass and grow in films attached to streambed rocks, leaves, and other substrates, creating a favored food item for many invertebrate primary consumers.

Beyond the major role that terrestrial production plays in subsidizing many river food webs, aquatic primary production is also an important and nutritious basal food resource. Given some light, carbon dioxide, and water, there will be algae – the main primary producers within stream ecosystems. One of the key invertebrate FFGs is the *scraper,* which obtains the majority of food energy by scraping algal growth from the surfaces of rocks. Scrapers include several species of mayflies, caddisflies, and true flies. Snails and some fish species also specialize on eating algae. These are integral components of river food webs which are themselves food items for various aquatic predators. These and other stream-rearing insects eventually emerge as flying adults, and although some of these return their carbon and nutrients to the aquatic ecosystem (e.g. as prey for fish, or by laying eggs into and then returning as dead organic material to the stream), the preponderance of emerging insects never return. Instead, they help fuel the terrestrial ecosystem. A number of terrestrial predators – including spiders, ground beetles, lizards, birds, and bats – organize their lifestyles and particularly their feeding strategies to take full advantage of emerging aquatic insects, and some birds (e.g. dippers, herons) feed almost exclusively on prey derived directly from the aquatic habitat, including fish. A guild of terrestrial invertebrate predators occupies riparian zones and feeds primarily on newly emerged insects still crawling across the aquatic/terrestrial ecotone. Bats will time dusk flights to coincide with heightened aquatic insect flight activity over stream corridors. And perhaps most eloquently: web diameter of some sheet weaver spiders (Linyphiidae) increases substantially with distance from the streambank. These spiders require only a very small web to be efficient at trapping the easy and abundant prey closest to the stream, but must weave a very large web to be fruitful at a distance. These and many other examples of the reliance of terrestrial predators on stream-derived prey have helped ecologists make a strong case over the past decade for the importance of aquatic subsidies to terrestrial food webs (e.g. Sabo and Power 2002).

Clearly, web connectivity across the terrestrial/aquatic ecotone has the potential to provide subsidies not only from more expansive terrestrial ecosystems to stream food webs, but also in the opposite direction (Figure 10.6). These so-called *reciprocal subsidies* are the norm in relatively unimpacted river ecosystems. The late ecologist Shigeru Nakano and his colleagues performed a series of powerful, classic studies of reciprocal subsidies of invertebrate prey to predators across the aquatic/terrestrial ecosystem boundary at a stream in Hokkaido, Japan. In one study (Nakano and Murakami 2001), these researchers showed that peak aquatic vs. terrestrial insect abundance varied temporally such that prey of stream origin subsidized terrestrial predators (primarily birds) substantially in spring, when terrestrial prey was in low abundance, and terrestrial prey subsidized stream predators (fish) primarily in summer, when aquatic invertebrate abundance reached a minimum. This bidirectional web connectivity therefore annually supported greater abundance of both aquatic and terrestrial predators than either ecosystem could support alone. In other studies, Nakano and colleagues experimentally tested the effect of severing web connectivity by erecting

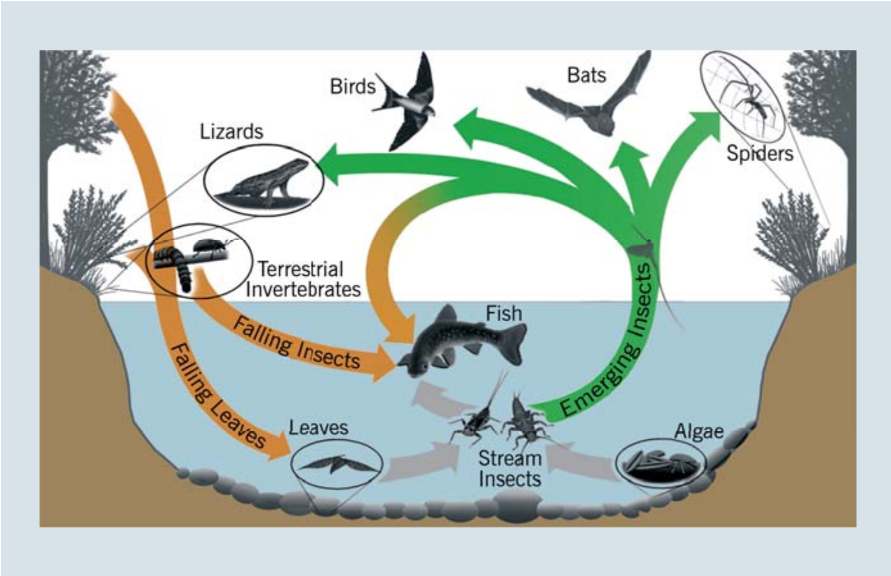


Figure 10.6: *Reciprocal subsidies. In streams, food web connectivity occurs in both directions: from terrestrial to aquatic ecosystems (trophic interactions shaded orange), and vice-versa (trophic interactions shaded green)*

a lengthy greenhouse-like structure of fine mesh directly over the stream (Figure 10.7) (e.g. Nakano et al. 1999; Baxter et al. 2004). The structure prevented most of the prey flux between the two ecosystems, with significant ecological effects in both, including altered feeding behavior and decreased growth rates of predators. Reciprocal subsidies between terrestrial and river ecosystems likely play a key role in overall ecosystem functioning in many river basins of the world. Unfortunately, many human activities, including those as disparate as channelization for flood control and introducing non-native sport fishes, can effectively sever aquatic/terrestrial connectivity – almost acting as a metaphorical *greenhouse* preventing reciprocal subsidies. We discuss several anthropogenic effects on connectivity below.

10.5.3. Marine/freshwater web connectivity

People commonly perceive rivers as one-way conduits moving materials from continents to oceans. An idea of streams as convenient sewage pipes of sorts led to a drastic increase in water pollution and associated environmental degradation during and following the industrial revolution. Currently, popular news stories about, for example, dead zones in near-shore marine ecosystems bolster the public's view of rivers as downstream conduits – and with good reason. Many marine dead zones are on the receiving end of rivers laden with excess nutrients (nitrogen and phosphorus in particular – often from agricultural activity in the catchments), and they reflect web connectivity between river and marine ecosystems. Excess river-transported nutrients that become available to marine food webs feed massive algal growth. Although algae produce oxygen in the presence of sunlight, they must respire it at night; so excessive algal biomass can result in the severe oxygen depletion characteristic of dead zones.

However, despite common perception of upstream-to-downstream unidirectionality of movement in rivers, marine ecosystems can also subsidize freshwaters via web connectivity in the opposite direction. A diversity of fauna worldwide is diadromous – a general term that describes animals that migrate at some point in the life cycle between marine and freshwater environments. Diadromous species include both fish (e.g. salmon, sturgeon, eel) and invertebrates (e.g. some crabs, shrimps, and snails). Longitudinal connectivity in river networks is required for these species to migrate, and a successful migration from ocean to upstream habitat results in movement of marine-derived nutrients and carbon between the two ecosystems. This transport of materials is of particular relevance in streams when it arrives via large-bodied anadromous species, like salmon. Anadromous species (a subset of diadromous) spend

Figure 10.7: *Nakano's "greenhouse" that severed lateral connectivity between stream and riparian forest*

most of their lives at sea where they feed and grow to maturity, then they return to fresh water to spawn. Spawning directly precedes death, which leaves carcasses – dense packages of nutrients and organic material, for all practical purposes – in and along stream channels and available to enter stream and terrestrial food webs via a number of pathways. In streams with strong spawning runs of salmon, researchers have shown that marine-derived nutrients can be found in nearly every node of stream and riparian food webs, from algae and microbes to predators and even to riparian plants and terrestrial consumers. Carbon and nutrients originating from the ocean and delivered in salmon carcasses subsidize stream food webs via direct consumption by animals, decomposition by fungal and bacterial microbes, and uptake of leached nutrients by primary producers. The additional nutrients and food energy that spawning salmon provide to stream food webs promote increased growth and reproduction of both producers and consumers compared to streams without spawning runs (Naiman et al. 2002).

10.6. Sustaining rivers as networks of webs: Conservation challenges

The term riverine landscape *implies a holistic geomorphic perspective of the extensive interconnected series of biotopes and environmental gradients that, with their biotic communities, constitute fluvial systems.*

J.V. Ward, 1998

The river is everywhere

Herman Hesse *(Siddhartha)*

10.6.1. Rivers as networks of webs

A synthesis that we have been converging upon is that river ecosystems in their natural state are fundamentally complex *networks of webs*. The example in the preceding section of salmon connecting marine and stream food webs via longitudinal connectivity in river networks reminds us that in fact these two concepts of connectivity are inextricably intertwined. *If there is no network connectivity, there can be no web connectivity*. In all river networks, dynamic webs of trophic connectivity take various forms in different localities along the longitudinal gradient from the myriad headwaters to main stem. Some webs (such as those linking marine and stream ecosystems through diadromous species) occupy a great deal of space and require extensive network connectivity in the longitudinal dimension to achieve trophic [web] connectivity. Other webs may be localized to a particular region of the network or may primarily depend on network connectivity in lateral or vertical – rather than longitudinal – dimensions. For example, aquatic/terrestrial web connectivity might be focused along a very short stream reach where ground beetles and wolf spiders patrol a small gravel bar, feasting on newly emerged stream insects. Alternatively, web connectivity can span multiple drainage basins, as when strong-flying caddisflies emerge in large numbers and disperse laterally, providing prey for forest birds far from the stream. The "riverine landscape" of J.V. Ward (quote above) embodies the concept of rivers as complex networks of webs, relying on connectivity in multiple dimensions and directions. More simply, as we read in *Siddhartha*, "the river is everywhere". A river is much more than a conduit from land to sea (Figure 10.8).

10.6.2. Anthropogenic impacts on connectivity

Rivers are essential to commerce, agriculture, transportation, and most other human enterprises – which causes immense pressure on river ecosystems, in-

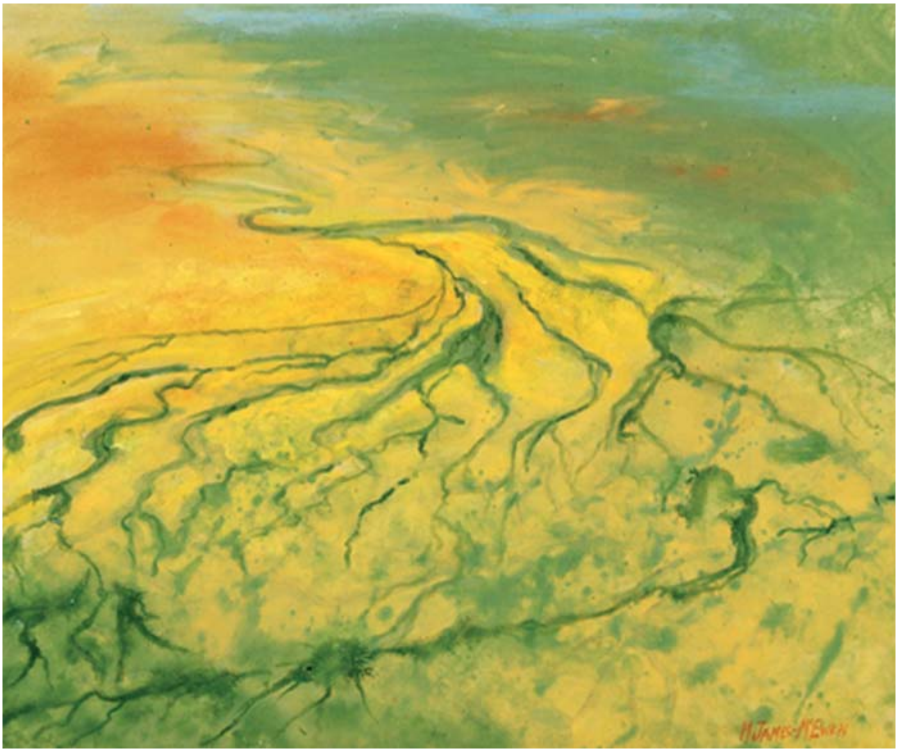


Figure 10.8: Green Fingers Across the Land, *a painting by Helen McEwen captures the idea of a desert river as a network of webs: the visible network structure (green) is emphasized by the lateral and aquatic/terrestrial web connectivity that results in riparian trees and other vegetation thriving along the riparian corridor*

cluding significant effects on network and web connectivity. Probably the most intuitive and frequently cited examples are dams. Even dams with fish ladders (which have variable success) or some other means to transport animals artificially around the structure disrupt longitudinal connectivity by altering the natural habitat gradient in a variety of ways; e.g. by creating a lengthy section of deep standing water (the reservoir) and impeding the downstream movement of nutrients and organic material (Ward and Stanford 1983). Dams also block the transport of sediments, resulting in profound effects on downstream habitat characteristics (Chapter 3). One function of many dams is to capture and store peak flows to avert downstream flooding. This thwarting of the annual flow pulse(s) decreases both lateral connectivity with floodplains and vertical connectivity with the hyporheic zone. Other dams act specifically to move water from the channel into ditches, canals, tunnels, or pipes for sometimes extensive transport to provide for irrigation or consumptive use in areas that do not themselves have a sufficient water supply for these purposes. These diversion dams often result in at least intermittent drying of downstream river sections, with clear negative impacts on connectivity in all three spatial dimensions. The assisted movement of water from one river network to the next, however, also creates an artificial *increase* in connectivity and can homogenize once-distinct populations and communities of strictly aquatic animals (Olden et al. 2004). This can result both from inter-basin transfers of water through canals and from long-distance transfer of water and organisms in the ballast water of ships, a major pathway for arrival of invasive aquatic colonists (Chapter 8).

Badly designed culverts under roads are another common disruption to longitudinal connectivity in smaller streams. Interestingly, stream sections upstream of perched culverts often contain fragments of native fish populations in regions where invasive species have been introduced in downstream reservoirs and larger rivers and spread throughout much of the network. The upstream-impassable culverts therefore represent the last bastions against invaders in many cases, leading to the perplexing management decision to leave the culverts unrestored in the interest of protecting these relict native populations.

Another pervasive engineering strategy in rivers is channelization, which typically involves straightening the course and installing riprap, levees, or even encasing the channel in concrete (e.g. the notorious Los Angeles River) in an attempt to prevent flooding in populated or agricultural areas (and sometimes to improve boat transport). Channelized rivers in general are almost completely disconnected in the lateral dimension, and concrete-lined rivers clearly also have zero vertical connectivity. The ecological effects of most channelized rivers can be anticipated from Nakano's *greenhouse* experiment that severed aquatic/terrestrial web connectivity, resulting in decreased productivity in both aquatic and terrestrial ecosystems. Furthermore, channelized rivers are disconnected from their floodplains, which historically would have provided nursery habitat and refuge from fast flows for aquatic animals, and highly productive riparian zones. The effects are large declines in aquatic and riparian biodiversity.

In anthropogenically modified river systems, restoration of web connectivity will often result indirectly from an emphasis on restoration of network connectivity

Changing terrestrial land use also affects both network and web connectivity in rivers. An extreme example is the practice of mountaintop removal coal mining, which literally removes mountain peaks or entire ridges to expose coal seams. The massive amounts of debris often get deposited in high valleys, resulting in burial of headwater streams and a total loss of connectivity of these important, capillary-like systems to terrestrial food webs and to the rest of the river network (Palmer et al. 2010). With any change in land use in a basin, even those less acute than mountaintop removal, it is important to consider both the impacts on cross-network connectivity via organism movement (e.g. the widespread mayflies now restricted to dispersal within riparian forest buffers, as above) and potential impacts of the changing terrestrial food web on local aquatic/terrestrial web connectivity (e.g. loss of riparian trees can lead to significant reductions in food web subsidies to the stream, both as leaves and prey items such as insects).

The burgeoning human population and our high demand for river-derived and other natural resources will continue to put pressure on river ecosystems and the connectivity necessary to sustain them. Reverting to near-pristine conditions is neither possible nor desirable. But there is hope for maintaining and/or restoring healthy levels of network and web connectivity hand-in-hand with river management for human needs. The final section of this chapter reflects on impacts, restoration, and continuing conservation goals in our home river network in western Oregon, USA, where we use various species of native fish as examples to illustrate connectivity issues in a densely populated river basin. Although a regional example, its real-world issues are representative of rivers everywhere, and we hope it will prompt readers to investigate connectivity in their own home networks.

10.6.3. Connectivity in our river: the Willamette

These tree trunks
These stream beds
Leave our bellies full

Portland, Oregon band The Decemberists (*Rise to Me*)

Dams and longitudinal connectivity

In the early days of human settlement in its fertile valley, the Willamette River provided the main north-south transportation corridor in western Oregon, USA. The Willamette is a major tributary of the Columbia River, and their confluence is just downstream of Portland, the largest city both on the Willamette and in the state. The main stem of the river, extending 290 km from the city of Eugene (second largest in the state) north to the Columbia, is wide and low-gradient, with only a single natural barrier to longitudinal connectivity (for human transport as well as some migrating animals): Willamette falls in the lower river network (Figure 10.9). Although the largest waterfall in the Pacific Northwest (by volume), Willamette falls is only 12 meters in height, and its cascading nature historically allowed longitudinal connectivity for three key diadromous fish: winter-run steelhead (the sea-run form of rainbow trout, *Oncorhynchus mykiss)*, spring-run Chinook salmon *(O. tshawytscha),* and Pacific lamprey *(Entosphenus tridentatus)*. The steelhead and Chinook runs were limited to winter and spring because winter/spring rainfall and spring snowmelt feeding headwaters in the Cascade mountain range drives a predictable flow pulse in the Willamette during these seasons. The high-water pulse allowed the spring- and winter-run salmonids to leap upstream through the hydraulics of boulders associated with the falls. Populations and species of salmon with summer/fall run timing, such as coho (*O. kisutch*) could not access the upstream network because of the seasonal constriction of the flow pulse, which made the falls a barrier to upstream connectivity for salmon during these drier times of year. Pacific lamprey have a different approach to breaching the falls and could do so even during lower flows by using primitive, jawless, suckerlike mouths to climb up the steep rocks of the cascade. Although not well known in the fish market, these eel-shaped fish (often called *eels* colloquially) are an important staple and ceremonial food for local Native American tribes.

A hydropower plant was installed at Willamette falls in the late 1800s, including a low, weir-like dam just above the cascade and small reservoir (Figure 10.9) – to provide electricity for the growing city of Portland. In spite of the dam, fish ladders have allowed successful passage of salmon around the falls and dam for at least the full history of the power plant. Indeed, the fish ladders have been *successful* to the extent that summer- and fall-run species (e.g. coho) and populations that historically were not present in the upper Willamette are now able to bypass the falls (although still in fewer numbers than their spring/winter counterparts) (Figure 10.10). Lamprey, conversely, do not use the fish ladders successfully. Instead, they appear to *climb* the falls as per usual but then are stymied by the vertical concrete dam. Furthermore, in the years since the Willamette falls dam and fish ladder were installed, a series of large, impassable dams have been constructed on most major tributaries upstream in the network – rivers that mostly occupy higher-gradient basins in the mountains surrounding the Willamette valley. Hence, although salmon now can readily bypass Willamette falls in the lower network, a large

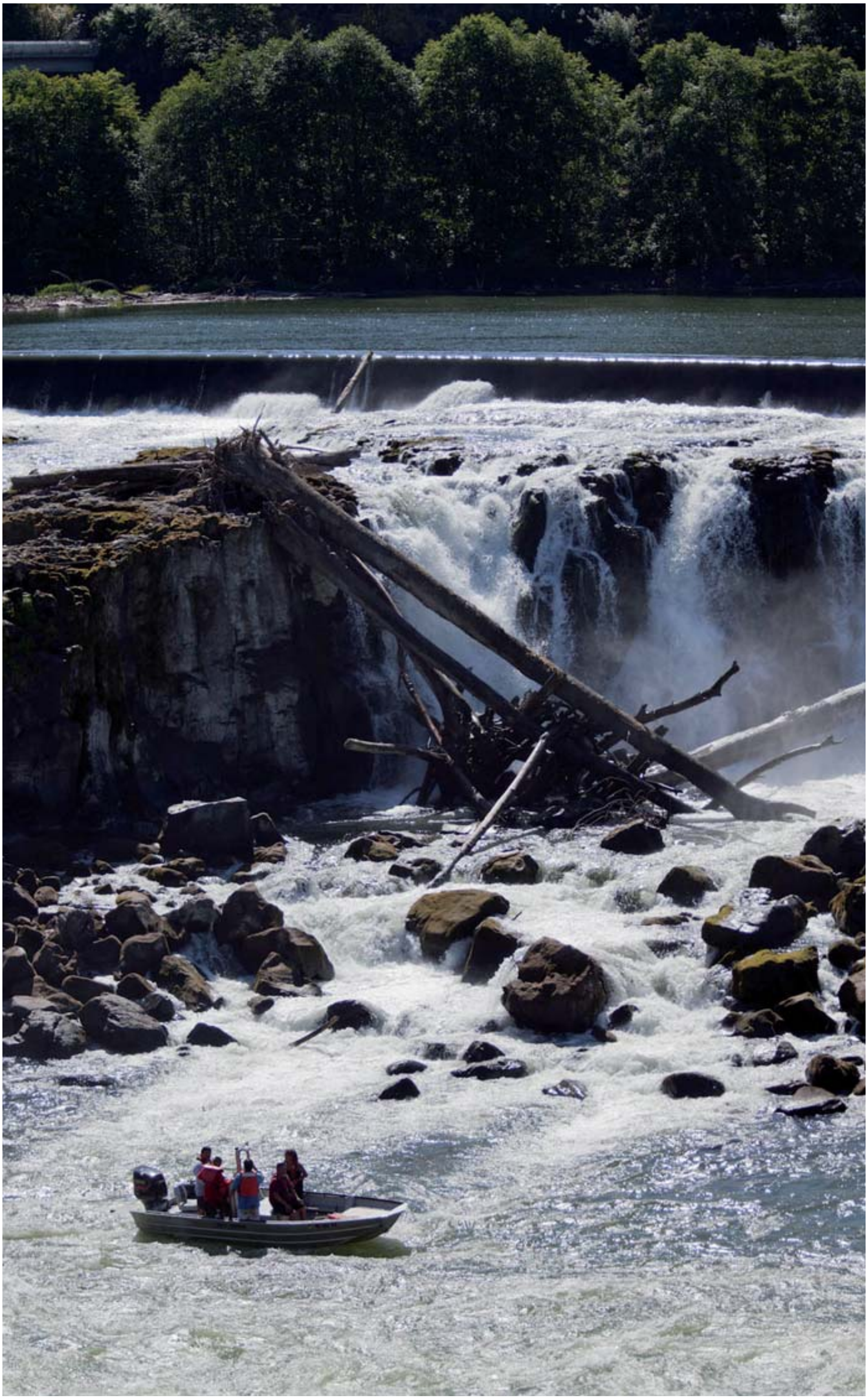
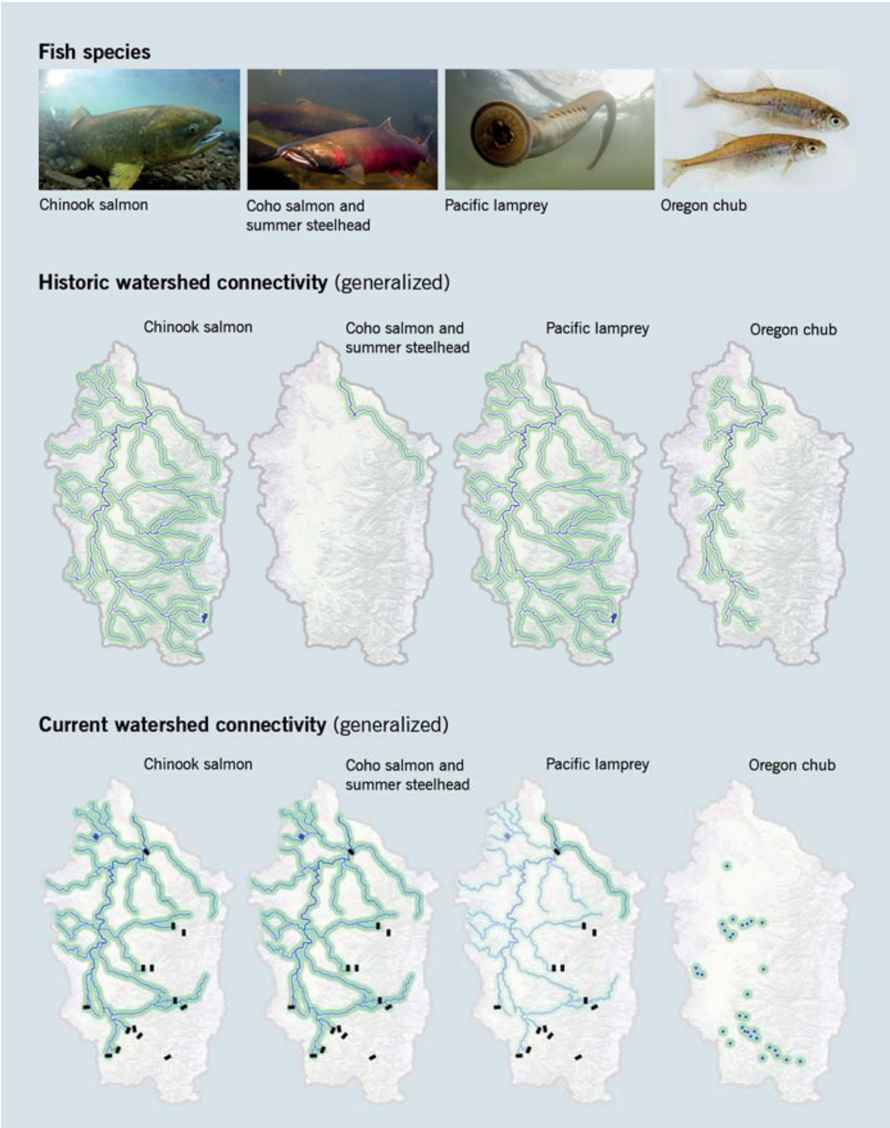


Figure 10.9:
Willamette falls and dam in the present day

proportion of their spawning habitat further upstream is inaccessible. The combined result is that both lamprey and the two native salmon populations are seriously threatened.

The good news for conservation is that restoration of longitudinal connectivity on the Willamette River is progressing. First, lamprey passage at Willamette

Figure 10.10: *Four groups of fish discussed in the text, and their historic (prior to human settlement) and current distributions in the Willamette River network, Oregon, USA. Black bars show approximate locations of dams; streamflow is from bottom (south) to top (north). The Chinook salmon represents diadromous salmonids with upstream spawning runs in winter or spring, when the natural flow pulse occurs in the Willamette. The coho and summer steelhead represent diadromous salmonids with spawning runs in summer or fall, when flow is too low to breach Willamette falls (near the site of the downstream-most dam located in the figures of the bottom panels), but the current fish ladder allows passage that was not possible historically*



Source: Hulse, Gregory and Baker (2002).

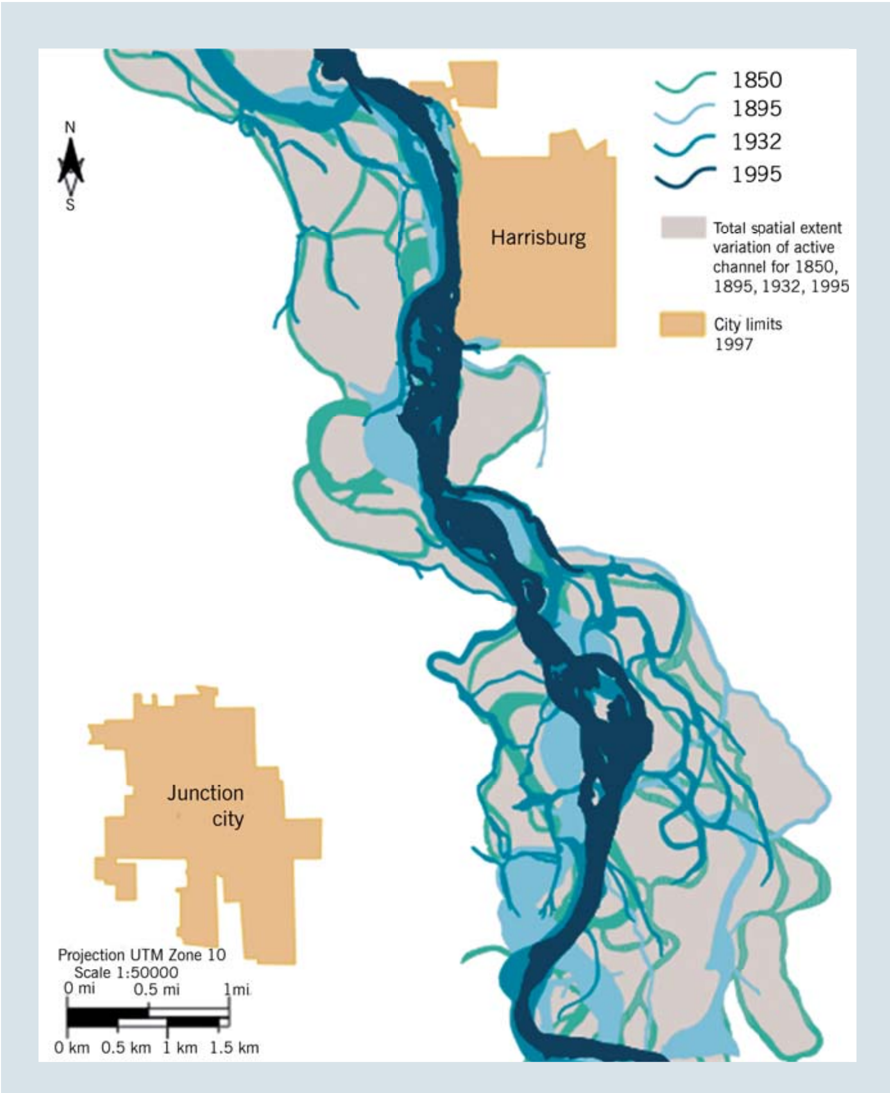
falls is a driving concern of state-level, tribal, and hydropower managers and is the topic of much ongoing research and monitoring. Installation of lamprey ramps (passage structures specific to the habits of lamprey) along the concrete lip of the weir, improvements to the fish ladder, and other efforts are underway and show promising results. Second, smaller dams obstructing tributaries and cutting off historic spawning reaches in the upper network are being removed, including recent removal of two dams on the Calapooia river. This Willamette tributary is now free-flowing, and longitudinal connectivity has been regained for spawning anadromous fish along its 99.7% of its length. Although larger dams on steeper tributaries in the Cascades are unlikely to be removed, there is a general consensus among stakeholders in the Willamette basin (unlike some other regions of the Western USA) that dams are a fundamental problem impeding recovery of threatened anadromous species, and concentrated efforts are underway involving diverse stakeholders to improve fish passage.

Channelization and lateral/vertical connectivity

Another pervasive impact on connectivity in the Willamette River has been channelization and loss of floodplain complexity in the period since settlement. As is typical of a natural river occupying a broad, low-gradient valley, the Willamette River mainstem until the mid-1850s boasted an a extensive and complex floodplain comprised of many braiding side-channels of varying permanence, islands, wetland, and riparian forest. Although some degree of this structure remains in patchy fragments along the river, the mainstem has lost around 2/3 of its total channel length (Figure 10.11) – and with it a significant degree of floodplain complexity and riparian forest – as a result of works to improve navigation and agriculture and to prevent flooding.

These changes represent a significant reduction in both lateral and vertical connectivity, and with it we can predict some loss of ecosystem function and ultimately decreasing aquatic and terrestrial biodiversity. But fish diversity remains relatively high in the Willamette, all things considered; and although nearly half of the list of current species is non-native, the data still reflect reasonable native fish diversity. Indeed, in the mainstem river channel 90-95% of the fish sampled by biologists are native, although non-natives increase in downstream reaches and in off-channel sloughs. Presumably, the essential functions of the intact floodplain for native fish would have been in providing refuge from winter/spring flood pulses and nursery habitats for developing young; and it turns out that an unlikely "new" habitat may to some degree be taking the place of the historic floodplain in providing these functions. To cope with the rainy winter and spring, most farmers in the Willamette valley run ditches through their fields to serve as drains. These ditches, though artificial, function as intermittent streams directly connected to the mainstem river, similar to small headwaters or to side channels in a floodplain. A recent ecological study of multiple agricultural ditches made four key discoveries: 1) many fish species occupy these habitats during the flood season; 2) the majority of these are native species; 3) many juveniles are present, indicating spawning and rearing; and 4) a good predictor of fish species diversity in a ditch is forest cover in its local catchment (Colvin et al. 2009). So, these

Figure 10.11: *Changes in Willamette River channel complexity and lateral connectivity from 1850 to 1995 in the vicinity of the towns of Harrisburg and Junction City (in upper mainstem of the river). Decreasing braiding, side channels and meandering has resulted in a loss of ca. 2/3 of total channel length since 1850*



Source: Hulse, Gregory and Baker (2002).

agricultural ditches are indeed acting as intermittent headwaters, and taking care of these ditches has become a new management directive. In the process, both farmers and native fish reap the benefits. These findings combined with a recent, multiple-stakeholder initiative to restore natural channel complexity and floodplain forest in promising locations along the mainstem Willamette allow cautious optimism for the future of lateral and vertical connectivity in our home river.

A connectivity vs. *invasive species conundrum*

The Oregon chub *(Oregonichthys crameri)* is a small minnow that loves pools found in murky, forested wetlands and is endemic to the Willamette River network. The chub probably once was distributed throughout the lowlands of the Willamette valley, associated with the complex floodplain aquatic habitat and riparian forests that have been so drastically reduced (Figure 10.10). It is now comprised of just a few isolated populations and is threatened with extinction. One of the key issues for the Oregon chub, aside from loss of its preferred habitat, is its inability to cope with invasive species. This situation leads to the conundrum (analogous to the case of relict native populations above dysfunctional road culverts) of how to manage this species concurrently with efforts to restore lateral connectivity. The problem is particularly perplexing because connectivity to the mainstem likely provides the key dispersal pathway for chub

Figure 10.12: *Agricultural ditch in the Willamette Valley draining productive cropland during the winter/spring rainy season (high-flow pulse)*

movement from one wetland habitat to the next (in the absence of the historic connectivity within the floodplain itself).

Concurrent with the push to increase lateral connectivity in the Willamette then, a series of management actions have been prescribed for the chub. The main priority is to protect extant populations from additional stressors (such as water extraction or chemical impacts, e.g. pesticides or herbicides from agricultural activities). In cases where ecological connectivity to the mainstem might be regained, fish barriers to prevent the influx of invasive species may be necessary. These will provide interesting case studies for monitoring habitats with restored ecological connectivity for essentially all functions aside from fish movement. Unfortunately, invasive bullfrogs *(Rana catesbeiana)* also negatively impact the Oregon chub, and – as we saw in the Necklace Lakes early in the chapter – barriers to fish connectivity are not necessarily barriers to amphibians. Bullfrog reduction is therefore another priority in managing chub populations. Another key management strategy will be to manually relocate individuals to other potentially suitable sites disconnected from the river network. And finally, protection and restoration of floodplain forests in key locations should provide an essential component of the Oregon chub's habitat requirements for the long term. Clearly, this problem, wryly nicknamed "chubs in tubs", is representative of the complex issues of multiple anthropogenic impacts, combined with the naturally complex ecology of river systems. (For more on the problem of invasive species in river ecosystems, see Chapter 8.)

10.7. Emerging concepts

This chapter serves the dual objective of first conceptualizing the interesting, complex, and necessary role of network and web connectivity in natural river ecosystems; and second moving to the "real world" where a multitude of river resources are necessary for modern-day human populations, but extracting those resources alters (sometimes severely) the natural connectivity so important for river ecosystem functioning. We converge on the idea that we can perhaps "have our cake and eat it too" by managing resource-extraction activities thoughtfully to maintain a reasonable representation of the complex network of webs characteristic of fully functioning river ecosystems.

A key to thoughtful management for connectivity will be to emphasize vital individual elements from the complex tangle of interactions that scientists understand natural river ecosystems to be. One such element is that natural flow pulses are essential for maintaining connectivity in all three spatial dimensions of river ecosystems. A flurry of research activity has occurred over the past ~15 years supporting the idea that what has been termed the "natural flow regime" (Poff et al. 1997) is a master driving variable in rivers. Maintaining components such as magnitude, timing, and frequency of flow pulses similar to what is expected naturally (e.g. given precipitation and snowmelt patterns) can preserve multidimensional ecological connectivity near natural levels even in highly regulated rivers.

Another vital element to emerge from the complexity is that *headwaters play a capillary-like role* in river networks by interacting intimately with the terrestrial environment and transmitting the effects of these interactions through the network. The same can be said for the small intermittent channels on floodplains. Taking care of these capillaries and their riparian areas should be a priority in holistic river conservation, likely resulting in a handsome return on investment as effects of healthy headwaters amplify through river networks.

Ultimately, it is important to remember that network connectivity in rivers must be maintained if we are interested in preserving web connectivity. The biotic interactions that drive web connectivity take place on the physical stage of network connectivity. Hence, what we might deem important and desirable outcomes of river ecosystem function (e.g. production of fish that we use for food, recreation, and ceremony) result *proximally* from web connectivity (e.g. the fish got enough to eat thanks to subsidies of terrestrial insects from the riparian forest) but *ultimately* from network connectivity (e.g. lateral connectivity between a river and an intact riparian zone). Analogously with the top Google definition of "connectivity", Internet Protocol is the physical connectivity that merely sets the stage for the more visceral connectivity of human interaction. What if we were to decide that sustaining connectivity along river networks is as important as sustaining internet connectivity?

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SCIENCE AND SOLUTIONS: MOVING BEYOND THREATS

CHAPTER 11

Ecological Restoration to Conserve and Recover River Ecosystem Services

MARGARET A. PALMER AND OWEN T. MCDONOUGH

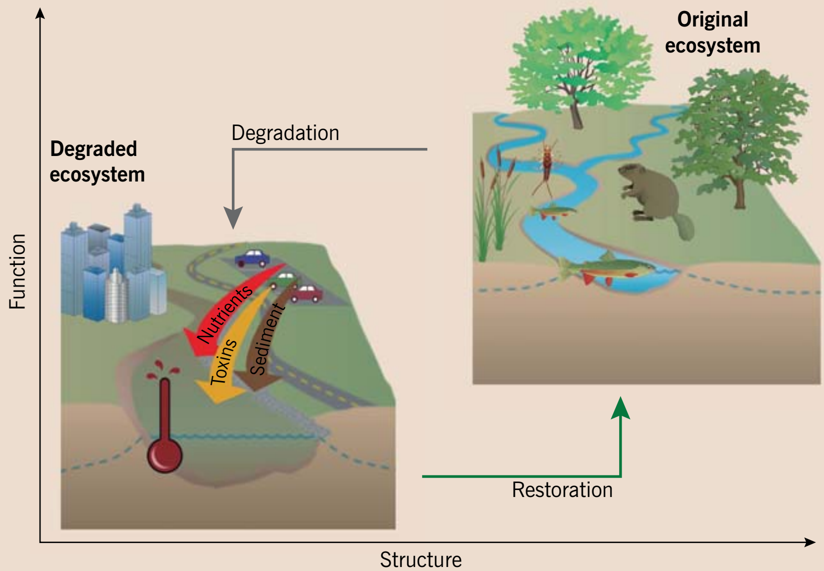
Ecological restoration of rivers and streams is increasingly shifting from a focus on reference sites to a focus on the conservation and recovery of ecosystem services that benefit humans. Strategies being employed to target specific biophysical features and processes necessary to support specific services range from simple interventions to ecologically designed solutions. The success of these restoration strategies often depends on broader catchment scale factors.

11.1. From restoring river ecosystems to restoring river ecosystem services

All living creatures depend on water for their very existence. Water is essential to basic metabolic functions, serves as a transport medium at scales from cells to biomes, and plays a critical role in global energy, mineral, and nutrient cycling. Despite this, hundreds of millions of people worldwide lack access to clean water. Most people rely on rivers for their domestic water needs as well as for irrigation, energy, and recreation. Humans also rely on the many goods freshwater ecosystems provide including flood protection offered by riparian wetlands and the source of food that fishery-rich rivers produce. However, there are many less obvious benefits that freshwater ecosystems provide such as water

Box 11.1

Ecologically successful river restoration



Source: Adapted from Bradshaw (1987). Symbols courtesy of the Integration and Application Network [ian.umces.edu/symbols/], Univ. of Maryland Center for Environ. Science.

Figure 11.1: *All ecosystems have two major attributes – structure and function.* ***Structures*** *are attributes related to the physical state of an ecosystem and are instantaneous measures; examples include population density, species richness and evenness, standing crop biomass, temperature, etc.* ***Functions*** *are physical, biological, and chemical processes occurring within ecosystems and often are expressed as rates; examples include biogeochemical cycles, production and respiration, accumulation and loss rates, population dynamics, etc. Structure and function can be used to illustrate ecosystem degradation. Though not always the case, the original ecosystem will be characterized by both high structure and function. Degradation decreases structure and function, whereas restoration attempts to increase both attributes in the direction of the original condition*

Humans have significantly modified the freshwater ecosystems on which we rely (Vitousek et al. 1997). Increasingly, river managers are turning to ecologically based restoration activities in order to improve degraded waterways. **Ecological restoration** is the attempt to return altered ecosystems to some historical condition (Box 11.1 Figure 11.1). Rivers integrate surface watersheds, ground-watersheds, and airsheds, and may arguably represent the most fundamentally altered ecosystems on Earth. In efforts to restore freshwater ecosystem goods and services, riverine and stream restoration have become both a world-wide phenomenon and a booming enterprise, with billions of dollars spent on restoration projects in the United States alone (Palmer et al. 2005). Yet, individual projects have been met with mixed success, and only recently have there been efforts to establish standards for what constitutes ecologically successful restoration.

Five criteria for ecologically successful river restoration (Palmer et al. 2005):

1. A guiding image for a healthy river must be identified *a priori*
2. The river's ecological condition must be measurably improved
3. The river must be more self-sustaining and resilient to perturbation
4. No lasting harm should be inflicted during construction
5. Pre- and post-monitoring must be conducted and data disseminated

purification, local temperature regulation, and carbon sequestration. Growing recognition that humans have and continue to seriously degrade the ecosystems upon which they depend has shifted public focus from a value- or aesthetically-based motivation to restore ecosystems to a *need*-based motivation (Palmer et al. 2004). This has had significant implications for how streams and rivers are restored, where restoration projects are implemented, and the directions restoration science has taken. We will elaborate on these, but first provide a brief overview of riverine ecosystem services and how they are linked to biophysical features within these ecosystems.

11.1.1. Restoration and ecosystem services

As river networks become increasingly human-dominated, restoration efforts will focus on the recovery of ecosystem goods and services upon which societies rely

Ecosystem goods and services are the outputs from natural systems that societies appreciate. They are benefits that people value and the reason investments are being made in river restoration. Society may be willing to pay for these outputs directly (monetary value) or their value may be quantifiable using non-monetary means (e.g. *relative valuation* where goods or services are compared and ranked). Ecosystem goods and services influence policies from regional to global levels, business transactions, and every day decisions by individuals (Figure 11.2). Ecosystem services are supported by a host of biophysical processes and ecosystem features. For example, abundant clean drinking water is an ecosystem service supported by many processes such as chemical transformations mediated by microbes and hydrologic fluxes including groundwater recharge and surface flows. This service is also supported by ecosystem features – types or components of riverine ecosystems such as vegetated riparian zones, hyporheic flowpaths, and floodplain wetlands (Chapter 9). It is important to recognize that the “products” of well-functioning and healthy rivers (in this example, clean water) are not equal to ecosystem goods and services. It is only when social value is placed on those products that they become goods and services. For instance, a healthy river that is inaccessible to people does not have social value unless individuals are willing to express a preference for preserving the existence of that river or retaining an option to use that river in the future or for future generations (Wainger and Mazzotta 2011).

Individuals rarely express preferences for the biophysical processes that underlie a riverine service (e.g. metal detoxification and organic matter decomposition may be necessary processes for the provision of clean water in some instances), thus we prefer not to adopt terminology that equates biophysical processes or ecosystem functions with services. We also think it is critical to distinguish biophysical processes and features from ecosystem services (Table 11.1) in order to emphasize the tremendous need to advance our understand-

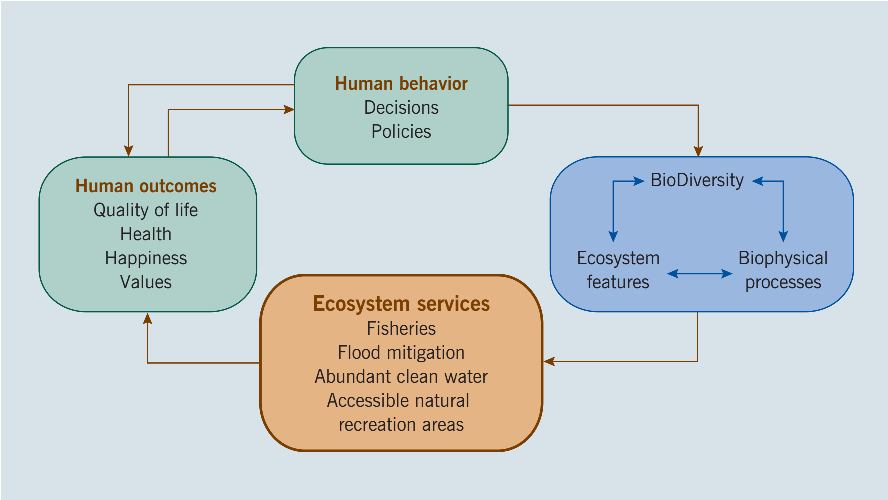


Figure 11.2: *Ecosystem services are the benefits people enjoy that come from natural systems. Their availability influences quality of life which is closely linked to human behaviors. Human behaviors, in turn, influence the components of natural systems: biodiversity, ecosystem features (e.g. different habitat types or structures at particular places), and a host of physical and ecological processes (e.g. water infiltration, nutrient cycling, primary production). Thus, the tight coupling between biophysical and social systems leads to complex dynamics for both humans and river ecosystems*

ing of when, where, and how those services are actually produced. While great progress has been made in identifying ecosystem services and developing methods for their economic or nonmarket valuation, the science behind which and what combinations of biophysical factors are essential to create and/or support these services is in its infancy.

In some instances, just a few processes may support a service, and in other cases, a multitude of complex processes interact to provide the basis for a service. For example, riverine flood control may depend almost exclusively on the presence of healthy, intact floodplains while productive riverine fisheries

Table 11.1: *Examples of riverine ecosystem services that people value and some biophysical processes and ecosystem features that contribute to the provision of those services. A few processes and structures are valued on their own and thus, depending on the context, could be considered services. Further, multiple processes and features may be linked to an individual service. This list is not intended to be comprehensive*

What people value (ecosystem services)	What makes those services possible (ecosystem processes and features)
— Clean water for drinking — Sufficient water at specific times for irrigation or hydropower generation — Flood protection — Food and food products (algae, rice, fish, invertebrates) — Recreation (fishing, swimming, water sports) — Aesthetics — Existence of species and ecosystems	— Nutrient cycling — Contaminant processing — Decomposition — Biodiversity — Water discharge and recharge — Heat and energy dissipation — Sediment transport and deposition — Riparian forests and wetlands — Floodplain connectivity — Channel form and woody debris

may depend on high rates of water infiltration in the catchment, a natural flow regime, intact riparian vegetation, and tight coupling of nutrient cycling with primary production. As such, depending on the ecosystem service a society wishes to promote, one or many processes and/or features may have to be conserved or restored.

11.2. River restoration goals

Throughout the remainder of this chapter we will discuss how restoration can work to create *potential* ecosystem services; i.e. the features and dynamic elements of a river ecosystem necessary to support a service. Riverine restoration should target those biophysical processes and ecosystem features most critical to the provision of desired ecosystem goods and services. As indicated in the prior section, the actual services assume there are social mechanisms or activities that ensure the delivery or availability of that service to people (Wainger and Boyd 2009). Quantitative relationships (i.e. equations or models) that allow us to predict potential ecosystem services as a function of biophysical processes and ecosystem features are the ecological or biophysical *production functions* underlying ecosystem service benefits.

To recover desired ecosystem goods and services, restoration actions should be guided by a scientific understanding of the mechanisms driving a river's ecological processes

For many decades, river and stream ecologists have worked to understand the factors that lead to ecological degradation and thus the need for restoration (Figure 11.3). They have also worked extensively to understand the relationships between physical processes such as discharge and sediment flux and important ecological processes and features such as rates of primary production (Young and Huryn 1996), decomposition (Webster et al. 1999), and biodiversity (Poff and Zimmerman 2010). In contrast, research on the relationship between restoration interventions and the recovery of physical and ecological processes in rivers is in its infancy. There is such a paucity of empirical data on the link between restoration outcomes and intervention practices that conservation biologists and natural resource managers largely rely on coarse-scale information based on correlations between human activities and river ecosystem degradation. For example, land use variables such as percent forest or impervious cover within a drainage basin serve as the basis for mapping the distribution of potential freshwater ecosystem services and identifying areas to be conserved or in need of restoration. Mapping services is valuable for guiding management focused on conserving parcels of land/water or on assessing the current status of services. However, mapping is typically insufficient to guide restoration actions because it does not provide ample *mechanistic understanding* (i.e. the scientific explanation behind a process) of the river and its processes.

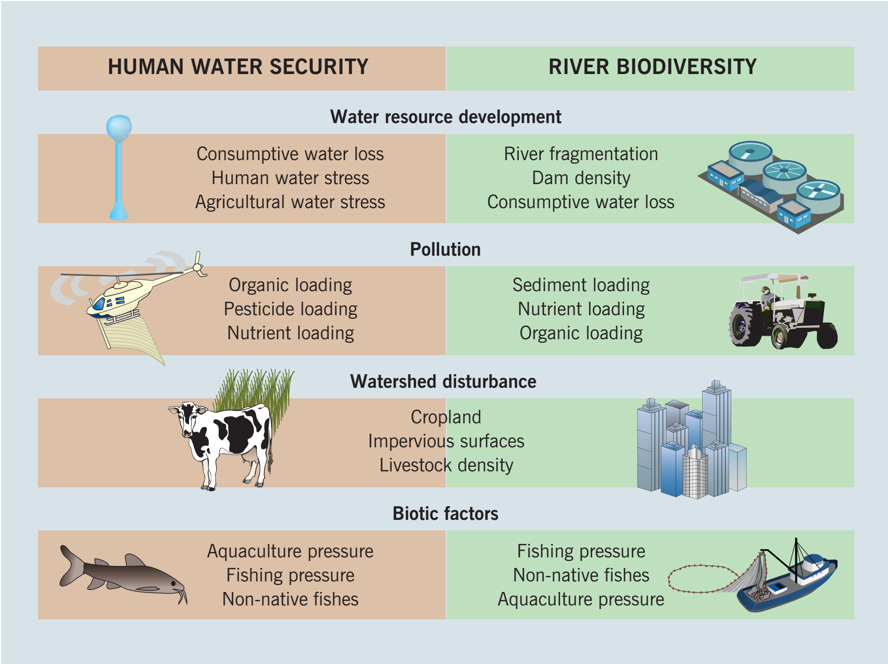


Figure 11.3: *Primary sources of river degradation that influence biodiversity* (right column) *and availability of water sufficient to ensure human well being* (left column). *Sources of increased impact are listed from top to bottom (i.e. the category "Water resource development" has the greatest impact among the four major categories, but within that category water consumptive losses have the most influence on water security, followed by human water stress and agricultural water stress)*

11.3. River restoration approaches

Selection of restoration approaches must be 1) based on a mechanistic understanding of ecological processes in rivers and 2) feasible from the perspective of managers. Correlational relationships may be adequate to predict *if* an ecological attribute is likely to exist in a particular location within a river network but not necessarily *why* or *how*. Sound restoration practices go much further because they involve hypothesizing the mechanistic links between the stressor (e.g. land use change, flow alteration, groundwater abstraction, etc.) and the state of the riverine attribute (Roni et al. 2011). These mechanisms are the key to identifying restoration interventions. For example, if we know that increased impervious surface causes increased overland flow volumes and velocities that in turn erode stream banks and incise channels, then we might target restoration efforts that reduce impervious cover within the catchment. Typically, our scientific knowledge of these mechanisms is based on data collected for systems that are being/have been degraded. But because the path to recovery may not mimic the path of degradation (i.e. *hysteresis;* Figure 11.4C), we cannot assume that quantitative relationships documented during degradation will hold post-restoration. For example, if biodiversity loss becomes significant only when certain stressor thresholds are exceeded (e.g. when impervious cover > 8-12%

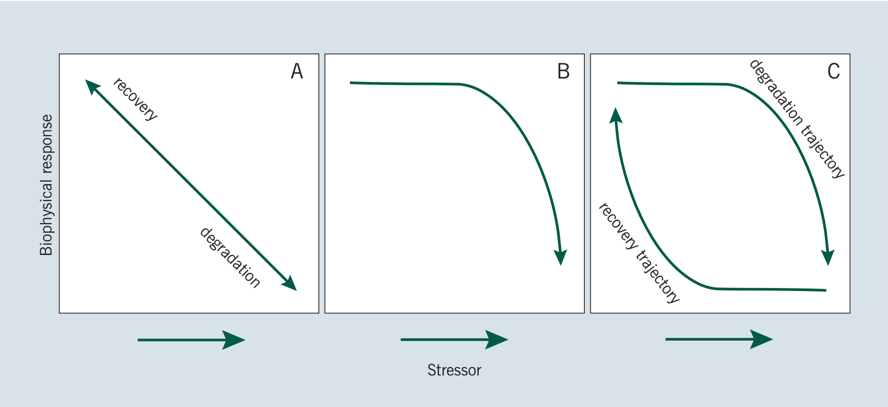


Figure 11.4: *River ecosystems respond in complex ways to stressors such as increasing levels of pollutants or uncontrolled flows due to land use change. The response may depend on the variable of interest or the context. For example, fish biodiversity may decline linearly as a stressor increases* (A – linear response), *or may remain relatively stable and only decline when a 'threshold' level of the stressor is reached* (B – threshold response). *A threshold response is particularly common when multiple stressors are acting simultaneously. Ideally, from a social and economic perspective, recovery is a direct response to restoration or management actions (as in panel A); however, many rivers exhibit a hysteresis response to disturbance such that recovery to former condition does not match the degradation trajectory and often involves a substantial lag time after the disturbance ceases* (C – hysteresis response)

[Stepenuck et al. 2002]; Figure 11.4B), does not mean biodiversity will recover if and when the stressor falls below that threshold. Hysteresis trajectories in environmental responses are quite common, and so for example, eutrophication in a river may not be reversed until nutrient levels are dramatically lower than they were at the onset of algal blooms (Duarte et al. 2009).

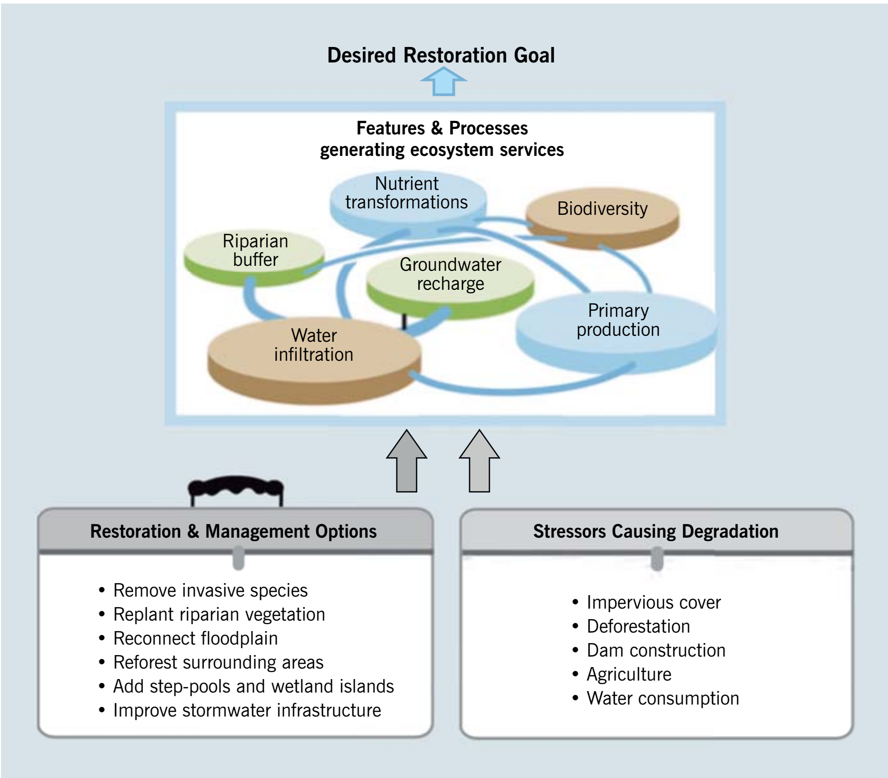
Correlational relationships are also often based on factors that catchment and river managers cannot influence. The "toolbox" from which managers can select when designing a restoration project may be limited by environmental policies and regulations, available funding, or social factors such as regional politics and land ownership. For example, as previously mentioned, there is a strong quantitative relationship between impervious cover and stream biodiversity, but managers are rarely able to remove all or most impervious cover in a catchment. Instead, they must focus on the fact that impervious cover limits water infiltration throughout the drainage basin which leads to a series of cascading events (e.g. rapid overland flow, bank erosion, channel incision, floodplain disconnection, groundwater table lowering, decreased base flow) that ultimately result in highly damaged waterways (Walsh et al. 2005a). Restoration efforts must focus on enhancing infiltration or some other intervention (e.g. decreasing overland flow velocities, armoring banks, re-connecting floodplains) that influences one of the other mechanistic paths that led to degradation.

As we discuss later, managers are typically tasked with implementing actions that will result in measurable benefits over small geographic scales and over short time periods. Their access to *intervention points* (i.e. where within the catchment they can implement restoration) is typically quite limited since most managers do not have policy controls that influence entire basins. In many cases, managers must understand where their tools can be effective at enhancing or restoring ecosystem services. To date, scientific research on restoration has rarely been based on starting with what tools managers and practitioners have available and where those tools can be used. Instead, most research and science-based prioritization schemes assume all options are on the table. An alternative and more realistic approach might be to ask 1) what options are possible, 2) what management/restoration tools are available, and 3) of those, which is likely to result in the greatest ecological benefits (Figure 11.5).

11.3.1. Restoration approach continuum: From conservation-based to technological approaches

Today, river restoration is practiced throughout the world and includes a diverse array of techniques that are often specific to a country or region. We can place projects into roughly four categories that vary with respect to the level of intervention (Figure 11.6). We can also characterize river restoration with respect to the broad goals that those funding or implementing projects hope to achieve (Table 11.2).

Figure 11.5: *Interventions used to restore river ecosystems must be based on the tools available to restoration practitioners and natural resource managers. Often, the stressful factors that cause river ecosystem degradation cannot be changed given the current socio-cultural context. Once the 'toolbox' of realistic options is identified, the interventions that are chosen (e.g. reconnecting floodplains or improving stormwater infrastructure) should be selected based on their ability to influence those ecosystem features or biophysical processes that are directly impacted by the degradation*



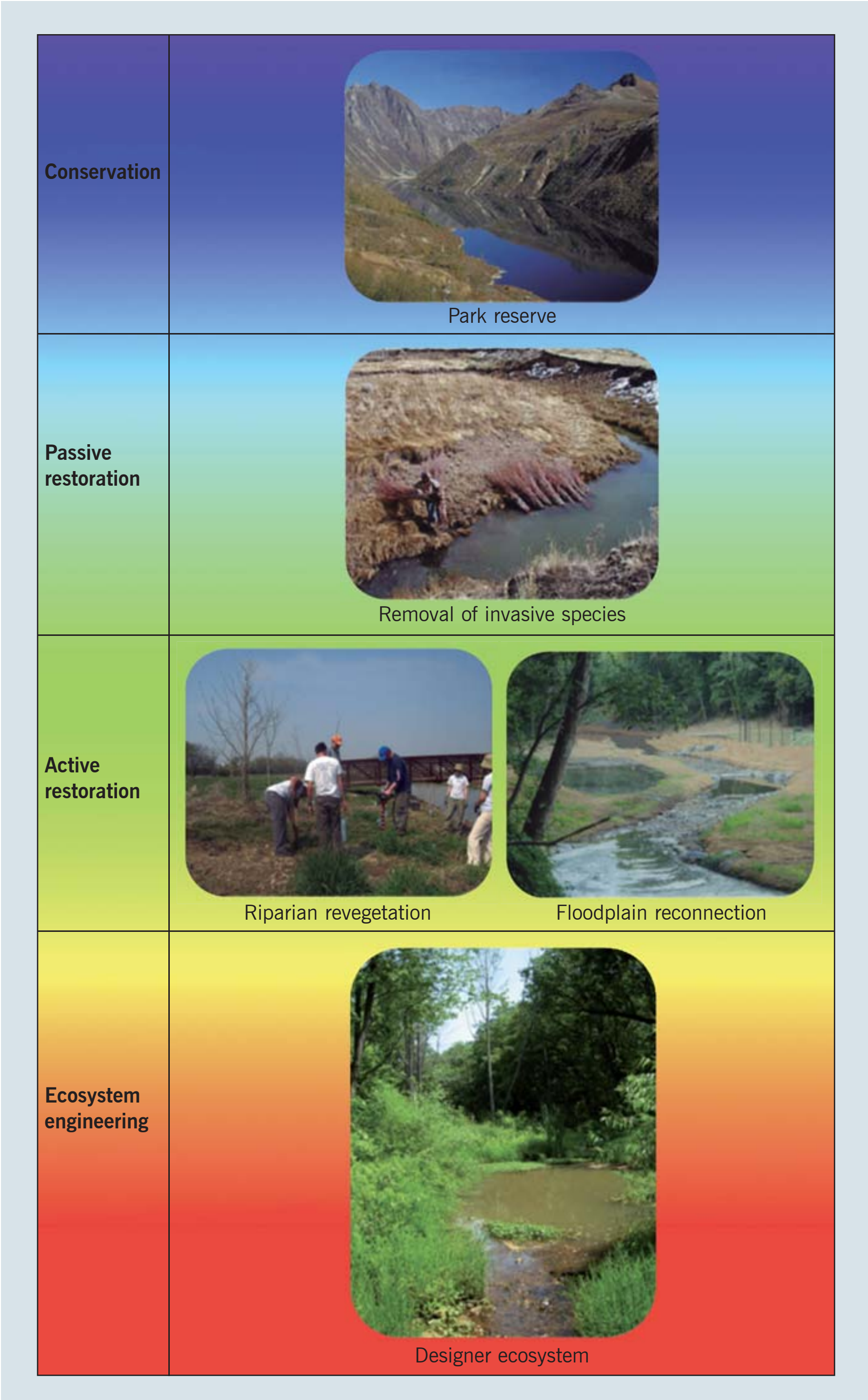


Figure 11.6: *Restoration of streams and rivers varies across a continuum from: simple* conservation *of land around a stream to protect it from expected degradation (e.g. due to encroaching urbanization) to* passive restoration *which occurs by natural processes alone after the major stressors are removed (here, invasive species were removed) to* active restoration *that involves various levels of intervention. The simplest intervention typically involves replanting vegetation along a river, but much more extensive forms of restoration are also common (e.g. bank armoring, bank grading, etc.). The end of the continuum is* ecosystem engineering, *the act of shaping ecosystems via active and passive means in order to provide desired ecosystem services. This may be accomplished by creating a 'hybrid' type ecosystem or an ecosystem type that might not be expected in a particular setting. Engineered channels are not actually restored streams and rivers since they do not conform to some past state or unimpacted reference site*

Table 11.2: *Common river restoration goals. Examples of common techniques used in river and stream restoration that may lead to ecological improvements. Most of these are part of an active restoration project. Each is based on a number of assumptions about the mechanistic link between the action and the desired goal. Qualitative 'scores' are provided to indicate the ecological effectiveness of each technique because there is generally insufficient empirical data to allow quantitative assessment of each technique's effectiveness in achieving desired goals*

Restoration goal	Specific actions	Mechanistic assumptions	Likelihood of success
Improve water quality	Planting riparian vegetation	Interception of overland flow reduces inputs of sediment and pollutants to stream	Moderate
	Soil conservation practices (e.g. no-till farming and cover cropping)	Increases water infiltration and reduces overland flow	High
	Livestock exclusion	Increases plant survival and stream bank integrity	High
	Control point source pollution	Eliminates pollutant inputs	High
	Bank stabilization	Reduces inputs of sediment from eroding banks	Moderate
	Reconfigure channels	Stabilizes stream bank, reduces erosion, enhances geomorphic complexity	Low
	Stormwater management	Reduces erosive urban flows and associated pollutants	Moderate for flow mgmt Low for water quality
Recover native species of interest or enhance biodiversity	Manually remove or kill non-native species; stock or re-plant natives	Natives will out-compete or prey on non-natives Natives will recover in the absence of non-natives	Low
	Enhance in-stream habitat (e.g. pool and riffle construction; addition of boulders or wood)	Habitat is the limiting factor, construction and structural additions will last, and desired species can colonize the river reach	Low
	Remove barriers to fish passage (e.g. fish ladder installation; culvert redesign; fish weirs on irrigation canals)	Passage is the factor limiting species recovery	High for passage Moderate for recovery
	Flow modifications (e.g. controlling the timing or magnitude of reservoir releases, limiting water extractions, adding in-stream flow diversions)	Water amount and/or timing of peak and low flows are primary factors governing species recovery	High if goal is to rewet dry streambed Low for recovery of species

Table 11.2: *(cont.)*

Restoration goal	Specific actions	Mechanistic assumptions	Likelihood of success
Recover basic river functionality	Daylight streams (i.e. redirection of a stream into an above-ground channel) Remove dams	Assumes ecological recovery will occur but time to recover depends on other sources of impairment	High for migratory fisheries in otherwise healthy catchment Limited information on recovery of ecosystem functions

High – strong empirical and/or qualitative evidence that technique is effective.
Moderate – may be effective depending on drainage basin context, exact design, and level of river degradation.
Low – reports of failure to see river improvements common.

Conservation of entire regions or habitat types associated with rivers is one of the most efficacious restoration approaches. Formal or informal policies that preserve riparian corridors or the headwaters of a river network are extremely important and effective means of restoring streams and rivers (Kline and Cahoon 2010). Protected parklands are particularly useful for conserving large tracks of land, while permanent conservation easements are good options if most of the catchment is privately owned. For the latter, a legal agreement between a landowner and government entity to restrict certain activities within a given distance from the river can promote recovery of healthy riparian corridors.

The natural – and often slow – recovery of rivers once a stressor is removed is called *passive restoration*. Putting impacted regions into conservation is certainly a form of passive restoration. Additionally, passive river and stream restoration is well documented when point source pollutant discharges are prevented, livestock are fenced out of streams, and water diversions and extractions are removed and/or prevented. This type of restoration can be remarkably effective for most streams but particularly those that are not severely or broadly impacted and those that have a high resilience capacity. For instance, rivers with an intact supply of colonists and within a catchment that has only a small area impacted will respond well compared to rivers that are highly degraded and more isolated from other healthy tributaries. Riparian corridors in grassland ecoregions that have been damaged by foraging livestock have been shown to recover quickly once livestock are excluded (Roni et al. 2002), and fish diversity can increase when barriers to upstream migration are removed (Gardner et al. 2011).

Active restoration in which streams, stream corridors, or in-stream biota or physical habitat are manipulated is assumed to be necessary in many cases – either because recovery is deemed unlikely without intervention or natural recovery would take an extreme length of time. The simplest, least expensive, and least interventionist form of active restoration is *riparian management*. This could include replanting vegetation along river corridors on agricultural or otherwise deforested land or controlling invasive plant species such as salt cedar *(Tamarix)* by manual or chemical removal. *Riparian revegetation* is among the most common restoration actions and is often combined with other active restoration approaches including bank grading, bank armoring, etc. It is important to note that while an intact riparian corridor is critical to ensure stream health, it is not sufficient – other factors such as urbanization in the catchment can override water quality or other benefits of riparian cover (Imberger et al. 2011).

In addition to simple interventionist techniques, removal of large flood and river control structures has become a common means to restore river function. Channel straightening and levee construction were historically assumed to reduce the risk of flood damage to property and human life along rivers and were thus extremely common forms of active restoration (Vitousek et al. 1997). Unfortunately, artificially straightened channels and levees may actually *increase* problems related to channel erosion and flooding (Gergel et al. 2002) both of which are expected to be even more common in regions predicted to experience higher flood magnitudes under future climate regimes (IPCC 2007). Additionally, flood control structures may actively disconnect rivers from floodplains, thereby impairing both running waters and their riparia. Removal or breaching of levees, therefore, is increasingly being considered to restore river and floodplain structure and function.

Similar to levee breaching/removal, dam removal has commonly been employed in efforts to restore natural flow regimes within river networks (Hart et al. 2002). While levees generally manage flow paths, dams serve the primary purpose of retaining water and, as a result, significantly alter natural flow regimes in rivers. As surface flow is a “master variable” in all streams, hydrologic modifications resulting from damming fundamentally alter both upstream and downstream ecosystem structure and function. While the long-term ecological benefits of dam removal can be substantial (e.g. restoration of natural flow regime, channel morphology, thermal regime, faunal dispersal), there may be adverse impacts immediately following removal. For instance, fine sediment transport following dam removal may adversely impact benthic habitat and deliver contaminants downstream (Hart et al. 2002). This suggests that from an ecosystem services perspective, societies may have to ask themselves which outcomes they most value with regard to the ecosystem in question and determine which available restoration option(s) would be most likely to produce those outcomes and over what time scales.

Restoration efforts aimed at improving water quality have also focused on floodplains as areas that slow flows thereby increasing interaction time between floodplain soils, microbes, and stream water. To date, however, few reach-scale studies that directly measure water quality benefits of *river-floodplain reconnection* have been completed, and those that have been conducted suggest only modest improvements in processes such as removal of excess nutrients (Roley et al. 2012). While decreasing flow velocities and increasing water-sediment interaction should in theory promote sediment trapping, nutrient retention/transformation, and channel stability, it is possible for catchment scale degradation to overwhelm any benefits derived from reach scale floodplain reconnection efforts (see below).

Among the most common forms of active restoration is *channel reconfiguration*. This can include a variety of actions (Figure 11.7) including but not limited to re-grading incised stream banks to reduce erosion, increasing channel sinuosity to slow flows, raising the channel bed to ensure floodplain connection during storms, and adding in-stream structures such as boulders or wood to provide additional habitat for biota and increase channel stability (FISRWG 1998, RRC 2002). In urban streams, restoration projects often focus on increasing channel

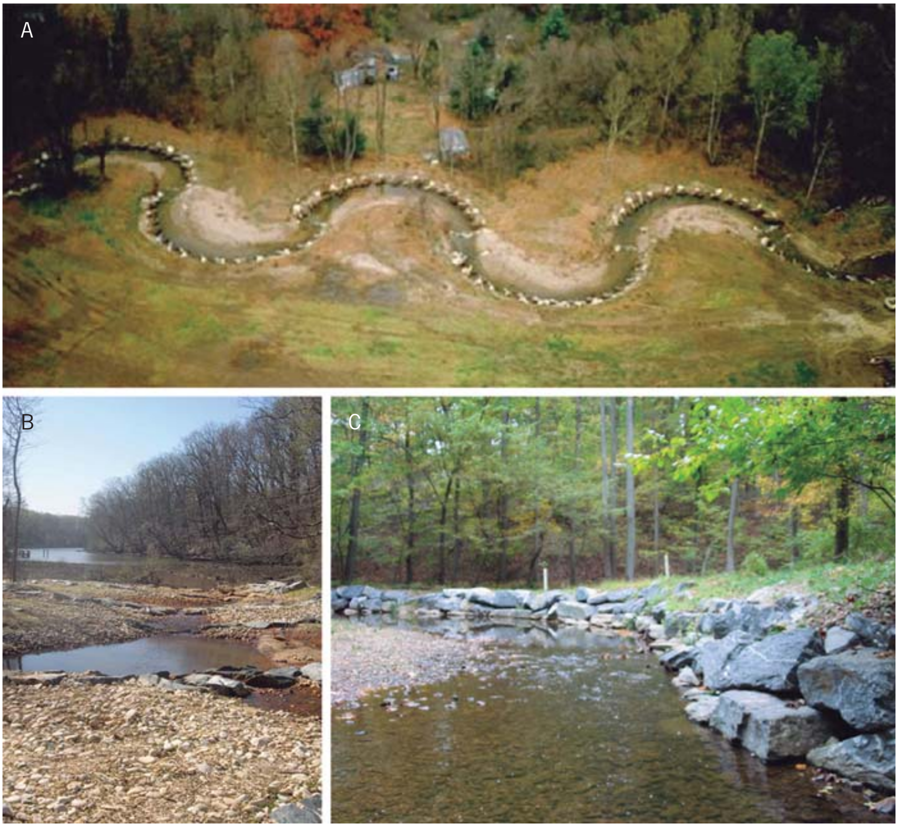


Figure 11.7:
Channel reconfiguration is a broad phrase used to describe a host of restoration projects that involve a range of earth-moving activities. In extreme cases, this might involve completely reshaping the channel dimensions (e.g. width, depth, sinuosity, etc.) as in panel A) It may also involve creating a series of step pools B) that sequentially reduce the stream power and erosion in streams that have been incised due to deforestation, agriculture, or urbanization. Many channel reconfiguration projects include detailed design plans to protect stream banks from erosion C) and/or provide potential habitat for stream biota. (All sites located in Maryland, USA)

Box 11.2

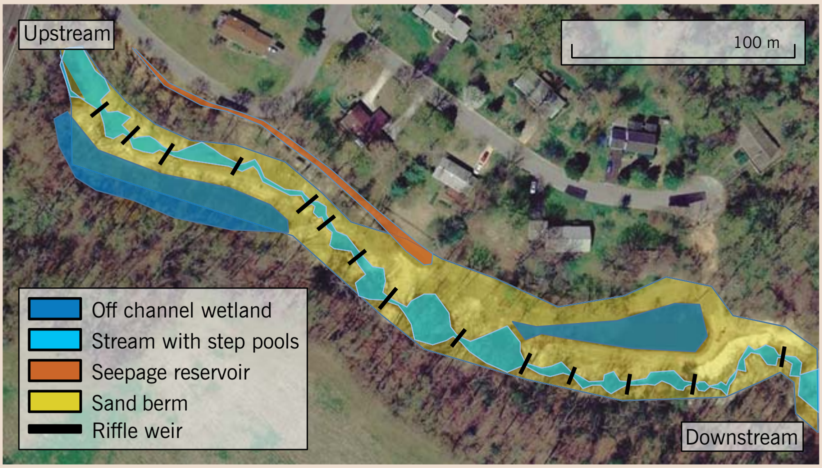
Wilelinor stream-seepage wetland: A case study in ecosystem engineering

With increasing societal demand for restoration of freshwater ecosystem goods and services, river managers and restoration practitioners are turning toward ecological engineering as a means of recovery. As an example, we highlight the Wilelinor stream-seepage wetland project – a "designer-ecosystem" recently implemented in a Coastal Plain tributary to Chesapeake Bay (Annapolis, Maryland, USA).

Problem: Urban development within the Wilelinor catchment has yielded significant sediment and nutrient loading to the stream and ultimately Chesapeake Bay. Additionally, stormwater velocities and peak flow volumes have increased due to nearly 40% imperviousness within the drainage basin. The Wilelinor stream was originally intended to provide recreational and aesthetic amenities to the surrounding communities. In recent decades, however, the stream and the benefits once enjoyed by local residents had become degraded as Wilelinor succumbed to the **"urban stream syndrome"** (see main text section IV). Residents voiced concerns with local government and demanded restoration of Wilelinor and other degraded waterways. In response to strong public interest in the restoration of recreational, aesthetic, and ecological resources, multiple state and county agencies collaborated to design a project with the goals of improving water quality and reducing peak flows and erosion.

Design approach: Rather than employing a traditional stream restoration approach (see Box 11.1), the agencies and practitioners incorporated multiple ecosystem design elements into the Wilelinor project (Figure 11.8). The result was a stream-seepage

Figure 11.8: *Wilelinor stream-seepage wetland site design, incorporating a combination of wetlands, step pool structures, sand berms, and weirs to slow storm velocities, promote floodplain wetland connectivity, increase hydraulic retention, reduce erosion, and improve water quality (38.967978 N, 76.544738 W; Annapolis, Maryland, USA)*



Source: Schematic adapted from Burke and Dunn (2010).

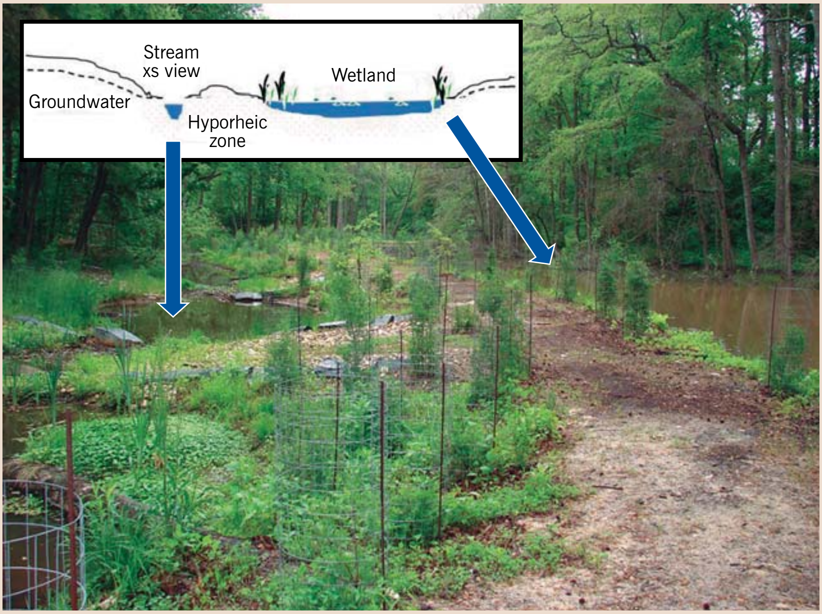


Figure 11.9:
Wilelinor is an ecologically engineered ecosystem combining both stream and wetland elements. The inset shows a cross-section schematic of the plan whereby the stream and wetland are hydrologically connected via overland flow during high discharge events and continuously via hyporheic flowpaths (i.e. below ground flow) through a porous sand berm

wetland hybrid (Figure 11.9). The design is intended to develop a stable stream profile and promote stream and floodplain wetland interaction, thereby slowing flows, reducing erosive power, and increasing hydraulic and nutrient retention.

Results: Since it was constructed in 2005, Wilelinor has been intensively monitored to assess the effectiveness of the design approach with respect to flow velocity and water quality restoration. Discharge data suggest that the stream-wetland complex effectively reduces peak flow velocity during storm events (Filoso and Palmer, unpub. data; Figure 11.10). Additionally, the system appears to be retaining nitrogen under average flow conditions and may significantly reduce N export relative to unrestored reaches (Filoso and Palmer 2011; Figure 11.11A). However, under high flow conditions, data suggest Wilelinor may not be as efficient at retaining N (Filoso and Palmer 2011; Figure 11.11B). The reduced efficiency of the system to process N under high flows is likely due to insufficient hydraulic retention and water-sediment interaction. Ongoing research is being conducted to understand the physical and biogeochemical factors governing nutrient and sediment dynamics within the stream-wetland complex. It is likely that stream-floodplain wetland interaction promoted by the project design plays a primary role in the observed reductions in peak flow and – at times – nutrient flux.

To effectively manage high nutrient and sediment loads and increase pollutant reduction capacity, streams may need to be increasingly manipulated or engineered, as

Box 11.2 *(cont.)*: Wilelinor stream-seepage wetland: A case study in ecosystem engineering

in the case of Wilelinor. It is likely that ecological engineering will play an increasingly large role in the recovery of freshwater ecosystem goods and services. Engineered ecosystems, however, may come at the expense of some portion of the fundamental structure and function of the original ecosystem (see Box 11.1).

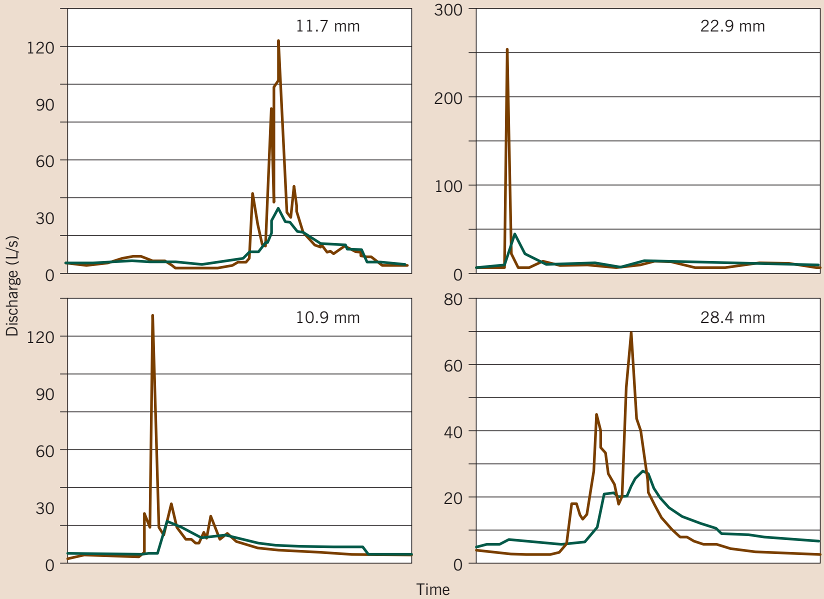


Figure 11.10: *Stream hydrographs of storm events of different sizes (rainfall in mm in each of the four insets) from discharge measured upstream* (brown) *and downstream* (green) *of the Wilelinor stream-seepage wetland system. As the four insets show, regardless of storm size, the magnitude and duration of peak stream flows were reduced*

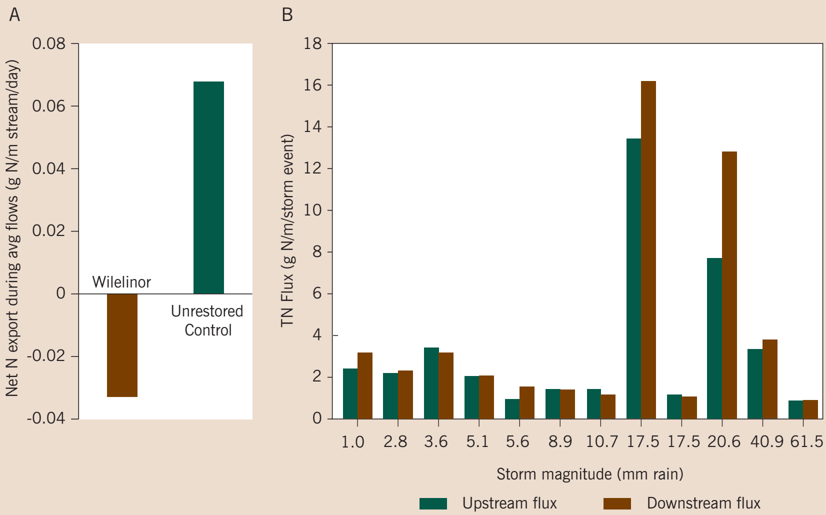


Figure 11.11: *A) Net nitrogen export during average flow conditions at the Wilelinor stream-seepage wetland and an unrestored control stream. Negative values indicate N retention. B) Total nitrogen (TN) flux upstream (green) and downstream (brown) of the Wilelinor stream-seepage wetland project during storm events of increasing magnitude*

Source: Adapted from Filoso and Palmer (2011).

stability and reducing erosive flows in hopes that in-stream biological recovery will follow (Niezgoda and Johnson 2005). Throughout the developed world, such channel-based or "hydromorphological" restoration projects are common, yet recent research efforts evaluating their effectiveness indicate that while they may stabilize banks and reduce erosion (Miller and Kochel 2010), they rarely lead to recovery of biodiversity (Palmer et al. 2010a).

Restoration in which all or part of the historic flow regime is recovered is not extremely common but represents a potential growth area as evidenced by the developing literature on *environmental flows* (Poff et al. 2010). The origins of environmental flow restoration are associated with streams and rivers in which flow diversions or extractions were sufficiently large that channels either ran dry for periods of the year in which they historically did not or water levels fell below those deemed sustainable for fish. In such instances the approach was to base flow allocations to rivers on information about the habitat needs for species of interest. Such restoration might require purchasing water rights or simply legislating minimum flow requirements. While environmental flows were originally based on minimum flow requirements, it is now widely recognized that natural variability in flow regimes is required to sustain freshwater ecosystems (Poff et al. 2010). With predicted increases in precipitation variability under future climate scenarios (IPCC 2007), environmental flow restoration is likely to be critical with respect to protecting aquatic species.

In the last decade, the concept of *ecologically engineered stream channels*, or "designer ecosystems", has sometimes led to projects that dramatically alter fluvial ecosystems – so much so that they can no longer be considered streams or rivers because they lack the geomorphic features and biodiversity characteristic of least disturbed or unimpacted reference streams in the region. Such projects typically involve a significant amount of earth-moving activity including for example, channel reconfiguration to create a wetland-stream complex which may also be connected to a stormwater reservoir of some type (e.g. Richardson et al. 2011). Step pools and in-channel sand berms may also be added to streams in efforts to enhance hydraulic retention and provide water quality benefits and habitat for wildlife. While this is often referred to as restoration, it is instead an attempt to recover specific ecosystem services using ecologically inspired approaches (Palmer and Filoso 2009). Ecological engineering is likely to play an increasingly important role in river conservation as societies shift from a focus on restoration of prototypic stream ecosystems to a focus on recovery of ecosystem services.

The concept of ecological design has been extended by some to include what is called *stream creation,* the attempt to construct a stream ecosystem where one did not previously exist. Stream creation is often confused with the common practice of *channel realignment* in which the position of a channel section or even entire reach is shifted laterally to conform to some historic condition or protect infrastructure along the channel that may be at risk due to erosion or flooding. Channel realignment is not stream creation because the river network and its longitudinal connectivity remain intact. Attempts to truly create a channel are typically proposed to mitigate for loss of stream resources due to anthropogenic activities including mining through or filling streams to extract coal or other valuable natural resources (Palmer et al. 2010b). There is no evidence that functioning streams can be created *de novo* as the few attempts thus far have failed to produce healthy streams with the full suite of ecological processes and native stream biodiversity (Palmer et al. 2010b).

11.4. Shifting restoration focus from the channel to the catchment

The vast majority of stream and river restoration projects are small in scale and isolated. Typically, individual reaches are restored, and often these are located downstream of smaller, degraded tributaries. Even when headwater tributaries are restored, if they are within a larger catchment with a high level of degradation, recovery may be minimal due to isolation from a healthy supply of plant or animal colonists. Stressors that lead to stream degradation are typically on a *catchment scale* – e.g. large amounts of impervious cover or land in agriculture. Commonly employed *reach scale* **r**estorations may be ineffective as they do not match the scale of degradation (Walsh et al. 2005b).

Despite widespread recognition that drainage basin and landscape context are critical to restoration effectiveness, only a small fraction of river restorations have been guided by a broader river or catchment management plan (Bernhardt et al. 2007). For most projects, sites are selected based on land availability. Problems stemming from opportunity-based site selection may be exacerbated if agencies and funders focus programs on specific habitat types, not broad regions (Palmer 2009). Further, regulatory frameworks may encourage small-scale, local interventions that fail to maintain the natural distribution of ecosystem goods and services. For example, under the U.S. Clean Water Act, mitigating for impacts to streams and rivers typically involves localized mitigation dictated by the amount of impact, and mitigation may result in significant spatial redistribution of freshwater resources (BenDor et al. 2009). In turn, mitigations for impacts to freshwater ecosystems may occur at significant distances from original impacts, and possibly in different drainage basins (BenDor et al. 2009). Reach scale restoration to offset impacts in a different catchment not only fails to restore structure and function within the impacted hydrological landscape, but will likely fail to yield ecosystem services equal to those lost.

When possible, restoration should be implemented at the catchment scale. Within channel reach-scale restorations are likely to be only locally and temporarily successful provided chronic drainage basin stressors are not alleviated. It is important to again recognize that managers are unlikely to have all possible restoration options and intervention points available on a catchment scale. Therefore, restoration should be approached by considering available options and tools and employing those most likely to produce the greatest ecological and/or socially valuable outcomes. For instance, managers working in urban catchments realize the importance of reducing impervious cover to alleviate the *"urban stream syndrome"* (i.e. ecological degradation of streams draining urban land and characterized by increased frequency of overland flow, increased nutrient and sediment loading, increased channel width and scour, decreased channel complexity, and decreased sensitive species [Walsh et al. 2005a]), but rarely have the power to remove all impervious cover within a catchment. However, as Walsh et al. (2005b) suggest, restoration may be used to decrease *effective impervious cover* within urban catchments (i.e. impervious surfaces directly connected to the stream by stormwater drainage infrastructure) and thereby efficiently target restoration efforts and maximize the likelihood of success.

Restoring river form may be insufficient to restore river function. A combination of conservation, restoration, and ecological design is needed to limit the loss of freshwater ecosystem services

11.5. Conclusions and recommendations

Over the course of our history, humans have modified the ecosystems on which we rely (Vitousek et al. 1997). Because river networks integrate surface watersheds, groundwater-sheds, and airsheds, they arguably represent the most fundamentally altered ecosystems on Earth. With human impacts often come the degradation of ecosystem structure and function. We are at a point at which restoring historic river form is likely insufficient to restore river function. As a result it is critical that we use a combination of conservation, restoration, and ecological design to limit further loss of freshwater ecosystem services.

We must recognize, however, that not all conservation, restoration, and design options or points of intervention will be available as we attempt to preserve the goods and services freshwater ecosystems provide. It will be increasingly necessary, therefore, to prioritize restoration efforts to maximize ecological and, in turn, societal benefits. We believe this can best be accomplished by approaching river restoration proactively at a catchment scale, rather than reactionarily using an isolated reach-by-reach approach.

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Chapter 12

The Role of Science in Planning, Policy and Conservation of River Ecosystems

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Richard Kingsford, Kim Jenkins and Fran Sheldon

So many of the big issues facing society are science-intensive, *and beneficial outcomes are unlikely unless science can be actively engaged in the development and assessment of appropriate policies. Climate change, over-allocation of water, endangered species issues as well as a raft of medical issues are all science-intensive issues where factual knowledge from science intersects with strongly held values.*

Peter Cullen, 2006

River conservation inevitably involves policy and planning with parties with disparate points of view. Successful river conservation needs informed science and the involvement of scientists. The California Delta and the Murray-Darling Basin are provided as current examples where science, policy, and planning are at the forefront for difficult river conservation decisions.

12.1. Science for river conservation

Rivers serve as the chief source of renewable freshwater for humans and contain some of the highest levels of biodiversity on Earth. Threats to rivers have become severe in many regions of the world for both securing human water supply needs and maintaining aquatic biodiversity (Vörösmarty et al. 2010). This dual challenge to rivers has resulted in nearly 80% of the world's population being exposed to high levels of threat to water security with 65% of riverine habitats classified as moderately to highly threatened. Changes in human population demographics and global economic activity in the coming decades will be predominant factors impacting future water supply and aquatic biodiversity (Vörösmarty et al. 2000). Threats to river biodiversity can be categorized into impacts from overexploitation, water pollution, flow modification, destruction or degradation of habitat, and invasion by non-native species (Dudgeon et al. 2006). Threats to water supply include water pollution, salinization, and human-induced climate change. Meeting ecological and societal needs for freshwater is one of the grand challenges of the 21st century (Jackson et al. 2001; Baron et al. 2002).

Jury and Vaux (2005) argue forcefully for the critical role science must play in addressing the world's water problems brought on by intensifying freshwater scarcities, growing populations, and developing economies. Two challenges for the effective use of science in water resource management are 1) applying contemporary and well-integrated knowledge of water resources in management and 2) planning and doing a better job of communicating with and educating water managers, decision makers, and the public. Water resources are often managed in a fragmented way (Jury and Vaux 2005). Examples include ignoring essential interrelatedness of ground and surface waters, failure to acknowledge crucial connections between water quality and water quantity, policies encouraging ground water overdraft, promoting short-sighted and wasteful agricultural water-use practices, and ignoring substantial benefits (ecosystem services) that flow from

Box 12.1

Planners, policy makers, politicians and decision makers

Four groups, along with scientists, play critical roles in determining the fate of rivers.

Planners coordinate diverse stakeholder groups to develop broad visions in the form of plans. They conduct qualitative and quantitative analyses and synthesize information to inform plan development.

Policy makers policies, which are purpose-driven courses of action. Public policy makers are generally government-appointed officials and may or may not be politicians.

Politicians determine policy decisions and are generally active in government. Politicians are often government policy makers.

Decision makers include managers charged with implementing projects, plans and policies. Policy makers also make decisions relevant to river management.

well managed and maintained ecosystems. Key to more successful management, planning, and communication is better understanding of the biological systems and processes that influence and are influenced by water availability. Extending scientific knowledge into the social sciences that consider human behaviour also is crucial for water resource management in the 21st century.

Likens (2010) asks, does evidence-based science drive environmental policy? This question is being tested in ongoing management decisions, planning efforts, and policy development for river conservation and restoration worldwide. Likens (2010) describes how human-accelerated environmental change requires better communication among scientists, decision makers, policy makers, the media, and the public. Long time periods may occur between detecting environmental problems and acting to alleviate those problems. Unassailable data, good communication skills, ethical integrity, the opportunity to communicate with planners, policy makers, politicians, and decision makers, knowledge of planning and policy, and perseverance are key attributes for effective scientific input into river conservation. Science can provide context and understanding, establish a framework for evidence-based policy and management, and guide the development of solutions through monitoring and synthesis, but science is not an absolute guarantee for understanding every impact of river restoration and conservation or a means to remove all uncertainty from the decision-making process. Science cannot solve all the problems in complex natural resource management challenges like the conservation and restoration of rivers, but science should provide the reliable knowledge base upon which decisions can be made. Good science, synthesized and interpreted well and communicated clearly, allows informed decisions by planners, policy makers, politicians, and decision makers.

There are numerous challenges for sustainable management of rivers throughout the world. In some regions, political paradigms are changing away from river development to river conservation and river restoration. In other regions, river development remains a central component of planning for feeding and clothing growing populations and providing power for emerging economies. In all cases, however, the role of science in setting policy, guiding planning, and influencing management is much debated and discussed. This chapter focuses on the role of science in policy, planning, and management of river ecosystems with a focus on our experiences in these arenas in the United States and Australia.

12.2. The policy, planning and management arenas

In the worlds of public policy, environmental planning and water management, there are many expectations for what roles science can fulfil and what expectations science can meet. These expectations vary based on the political climate, including different and often competing government agencies, the spatial and temporal scales of the river resources and the complexity of the issues. Science is often inserted into the policy, planning and management arenas through major policy decisions (i.e. mandates or laws), regional planning decisions and management actions that can range in scale from local to national. Often, the role of scientists is managing expectations from planners, policy makers, politicians, and decision makers as to the extent and way science can assist decisions given available data and defining the scope of the problems to be addressed.

Science can be inserted into major policy decisions through national or regional government mandates for the purposes of advising and informing decisions about complex social-ecological systems. These decisions can sometimes result in political and policy actions. Politicians and government officials may utilize science to provide evidence for action and guidance for preventing or rehabilitating a problem. However, creative tension often exists between scientific viewpoints or interpretations and the rationale for action. Around the world, science informs major policy, required by law (e.g. Sullivan et al. 2006a,b; Ryder et al. 2010). Several mandates specifically require the use of "best available science" (BAS) (see Box 12.2).

United States federal law in the mid-1960s first required that science guide decision-making for natural resources management decisions, and the Endangered Species Conservation Act in 1969 imposed a requirement that the "best available scientific and commercial data" be used in listing endangered species (Figure 12.1). In Australia, national policy also requires the use of "best availa-

Box 12.2

"Best available science"

The term "best available science" (BAS) is used widely in national, state, and local policies around the world. Its definition continues to be debated among scientists and decision makers and has become a premise for litigation. Several efforts exist to develop criteria for best available science. In 2004, there was acknowledgement that guidelines and criteria must be defined for best available science in natural resource management in the United States (National Research Council 2004). Recommendations included establishing procedural and implementation guidelines to govern the production and use of scientific information. These guidelines were based on six broad criteria: 1) relevance, 2) inclusiveness, 3) objectivity, 4) transparency and openness, 5) timeliness, and 6) peer review.

ble science". The Australian Water Act of 2007 that created the Murray-Darling Basin Authority (MDBA) and specified the development of a Basin Plan for the sustainable management of the Murray-Darling Basin states, "the Authority is to determine the volume of water required to maintain and restore environmental assets and functions, using best available science and the principles of ecologically sustainable development".

Planning efforts for rivers (i.e. balancing river water supplies for human and ecosystem needs) are often a product of disparate legislation required to manage a variety of natural resources. Multiple plans are often developed (particularly when multiple agencies are involved) that conflict with little thought on how to reconcile the multiple goals of the various plans. Complex and competing demands on river resources provide excellent examples where competing planning efforts aimed at maximizing river resources for human and ecosystem needs are developed with competing goals, objectives, and assumptions (see examples below). This leads to problems of planning integration and issues of

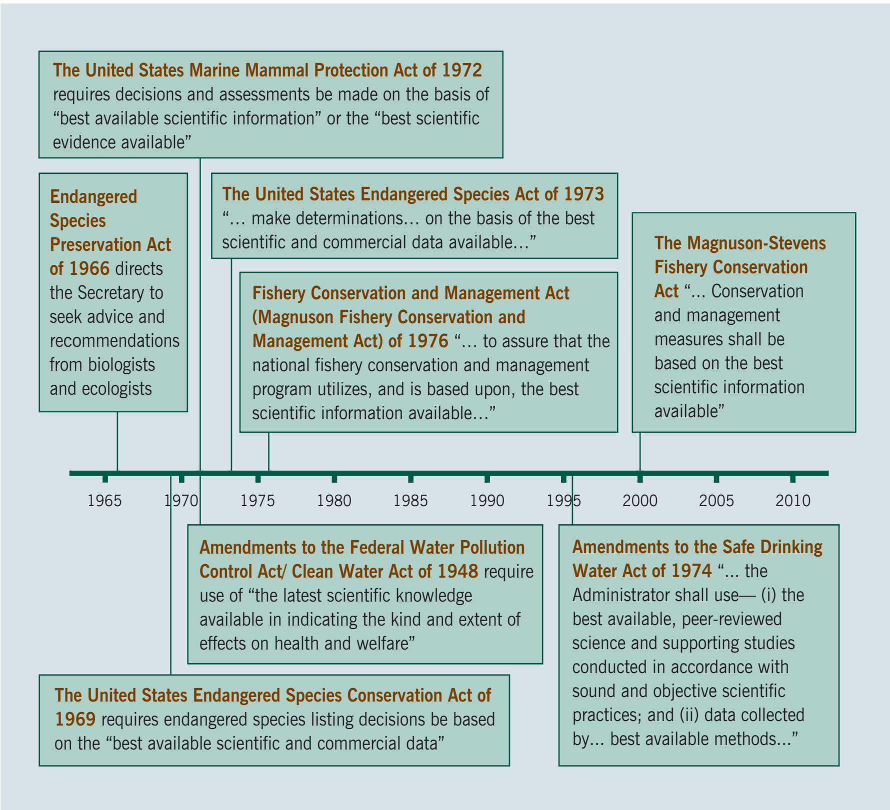

Figure 12.1:
A chronological summary of science requirements found in major environmental laws in the United States

scale and complexity in dealing with uncertainty. Science is often inserted into these planning efforts in order to assess the sources and magnitude of the uncertainty when dealing with complex and competing goals. Environmental policy is most effective when scientific uncertainty is incorporated into the decision process as *knowledge* rather than ignorance (Bradshaw and Borchers 2000); this helps policy makers assess where uncertainty lies and its seriousness. For example, numeric values can be presented as ranges of likely values, and assessments and conclusions can be rated as to the degree of certainty (e.g. high, moderate, or low). In general, uncertainty increases with increasing complexity, spatial and temporal scale and system variability – all features of most river basins.

Science also can inform conceptual models that provide a rationale for selecting plans and actions likely to achieve their intended goals. An excellent example for the use of conceptual models in river conservation and restoration is the South-East Queensland Environmental Health Monitoring Programme (EHMP) in Australia (Bunn et al. 2010). EHMP uses conceptual models and objective testing utilizing 16 indicator metrics to diagnose probable causes of river degradation arising from multiple stressors. The approach taken in this programme leads to more targeted management for river conservation and rehabilitation. Key lessons from this successful programme include the importance of a shared common vision, the involvement of committed individuals, a cooperative approach to problem solving, and defensible science with effective communication.

Managing the resources of rivers, through the implementation of plans, programs and projects, involves regularly confronting uncertainties. These uncertainties are inherent when managing complex systems. One role of science in management is to help define a process for acting under uncertain conditions (Likens 2010). These processes include Strategic Adaptive Management (Chapter 13) which includes targeted research to address specific objectives and uncertainties, monitoring feedback loops, and synthesizing current understanding for improving future management actions. Science also plays a key role in building tools (e.g. conceptual models, predictive models, scenario testing, and decision support systems) for guiding management decisions under uncertainty (Bradshaw and Borchers 2000). Uncertainty is commonplace in complex human and natural systems such as the economy, public health, and climate change and is not simply the domain of water resource management or environmental issues.

12.3. Inserting science into policy, planning and management

Insertion of science into policy, planning and management is essential for informed decision making but can yield both positive and negative results. While providing objective information, science can become value-laden in how it is inserted into decision-making. This can lead to science being used to promote one agenda over other competing agendas ("combat science") and as grounds for litigation (Hanak et al. 2011). How, when and what science is used to inform policy, planning and management decisions affects the perception and reality of how well science helped to inform the actions taken and its value in decision making. This is not an argument against the use of science in decision making concerning such challenging topics as river conservation and restoration but a caution that how science is summarized, packaged and communicated affects how science is used for making management decisions.

High value is normally placed on the quality of the science used to inform policy, planning and management decisions (Box 12.2). Independent scientific peer review helps ensure that best available scientific knowledge for decision or policy making processes is applied in an objective, transparent and scientifically valid manner, especially when the decisions are controversial or associated with high uncertainty (Meffe et al. 1998). Independent open review of programs, plans and products to promote the use of best available science in policy, planning and management enhances the chances that high quality science will be incorporated into decision making. Monitoring and evaluation also provide objective scientific support for decision makers. These programs build the scientific knowledge base to answer complex questions in river policy, planning and management. Coupling a strong monitoring program with a well-designed synthesis and integration effort (e.g. Strategic Adaptive Management, Chapter 13) improves the likelihood that high quality science will inform policy, planning and management of river ecosystems.

We have had considerable experience in working at the interface between science and policy for the management of rivers in two high-profile regions in the United States and Australia. The California Delta is the confluence of the Sacramento river and the San Joaquin river, and the delta is the heart of the largest water supply system in the world. The Murray-Darling Basin is the focus of substantial agricultural production in Australia and often called the "bread basket" of Australia. We focus on the role of science in planning, policy and management for these two catchments while acknowledging similar challenges in riverine landscapes worldwide.

12.4. Science and policy in the California Delta

The California Bay-delta (Delta) catchments encompass about 40% of California and the catchments for the rivers receive about 50% of the annual precipitation that the state of California receives. The Delta is one prominent example where balancing water supply needs and sustaining biodiversity is difficult (Figure 12.2). The Delta is the heart of the largest water supply system in the world (Dahm 2010). Precipitation in northern California and the Sierra Nevada flows into the Delta, and some of this water is pumped from the Delta by two large pumping facilities for use by urban and agricultural areas of central and southern California. The Delta ecosystem provides some of the water supply for ~25 million Californian residents, irrigates about one million hectares of farmland that accounts for ~45% of the fruits and vegetables grown in the United States, and is home to ~50 species of threatened or endangered plants and animals. The Delta also supports a local rural economy and is home to about half a million people.

Approximately 80% of the water flowing into the Delta derives from the Sacramento River, 15% from the San Joaquin River, and 5% from rivers that enter the

Figure 12.2: *Part of the California Delta, confluence of the Sacramento and San Joaquin rivers*

Delta from the east (Figure 12.3). Water quality is variable in the various source waters with generally higher water quality coming from the Sacramento River and the eastern rivers than from the San Joaquin where agricultural runoff dominates flows during much of the year. Interannual variability in precipitation in California is the highest of any state in the United States (Dettinger et al. 2011). This leads to highly variable natural flows in the rivers, and significant total annual precipitation derives from intense brief oceanic storms ("atmospheric rivers") sweeping in from the subtropical Pacific Ocean. Therefore, water supply is strongly linked to floods in many rivers of California and commonly comes with a few high intensity storms. This highly variable supply of precipitation and spatially variable distribution of water has been the impetus for water works that reallocate and export water from the Delta. Exports from the Delta have increased from around 1,200 gigalitres (GL) in the 1940s to ~6,200-7,400 GL in recent decades (Culberson et al. 2008). One gigalitre is the same as one cubic hectometre, one million cubic meters, or 811 acre feet. Conservation planning for the Delta focuses on 1) water exports (amount, timing of withdrawal, hydrodynamic impacts, and effects on water quality), 2) the most effective way to convey water (through, around, or beneath the Delta), and 3) river and marsh restoration.

Critical issues and drivers of change in the current Delta include climate variability, water quality, land subsidence, sea-level rise, earthquakes, invasive spe-

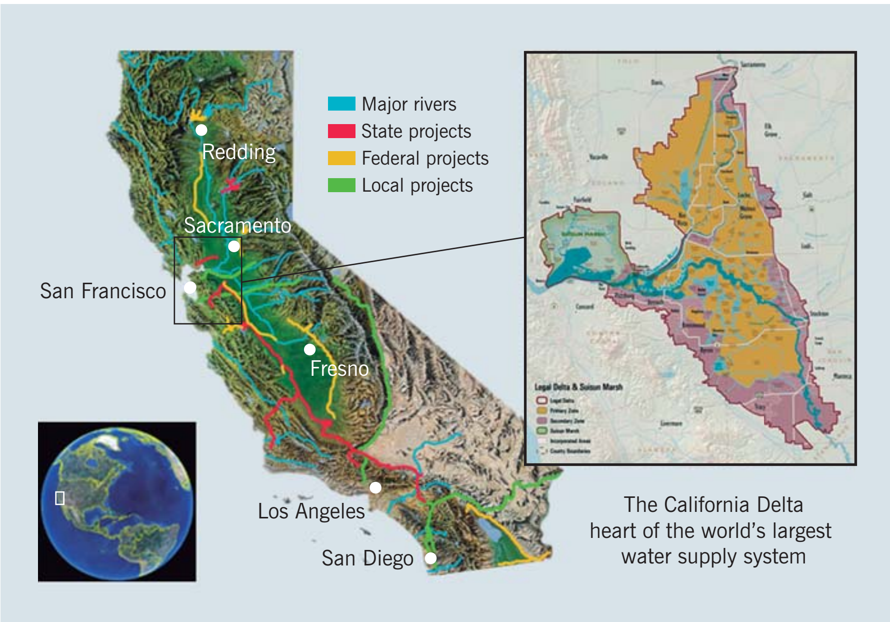

Figure 12.3: *Location of the California Delta at the confluence of the Sacramento and San Joaquin rivers*

cies, human population growth, and climate change. For example, the islands of the central Delta have subsided up to nine meters below sea level, and they are threatened by catastrophic flooding from sea level rise and earthquakes. Climate change scenarios for this century for the basin predict warmer temperatures of 1.5 to 4.5 °C, a one-third loss of snowpack in the Sierra Nevada by 2050, and higher and flashier winter river flows and lower summer flows with longer periods of low flows (Cayan et al. 2008). The recent dramatic decline of open water (pelagic) fish species in the Delta has drawn political interest and spurred considerable research on water movement, food webs, nutrients, contaminants, and habitat. This scientific research is now being incorporated into major planning documents to guide restoration of key attributes of the Delta while maintaining needed water supplies for California.

12.5. The Delta Reform Act of 2009

Recognition of the declining condition of the Delta and the need for increased reliability of water supply culminated in new State of California legislation (November 2009) aimed at addressing these dual challenges. The Delta Stewardship Council (Council) was created through this legislation to achieve the state mandated "coequal goals" for the Delta. "Coequal goals" is defined by state statute as the two goals of "providing a more reliable water supply for California and protecting, restoring, and enhancing the Delta ecosystem." In addition, the statute requires that "the coequal goals shall be achieved in a manner that protects and enhances the unique cultural, recreational, natural resource, and agricultural values of the Delta as an evolving place."

The Delta Reform Act of 2009 (Act) established statutes for the role of science in the California Delta. A Delta Independent Science Board with up to ten members was established to provide scientific oversight for research in the Delta. The Delta Science Program was placed under the Council with a vision that Delta water and environmental policy is founded on the highest calibre science and a mission to provide the best possible scientific information for water and environmental decision-making in the California Delta. This is to be accomplished through supporting research, synthesizing scientific information, facilitating independent peer review, coordinating science activities, and communicating science. Statute also requires that adaptive management be used in decision-making and developing policy. The Act defines adaptive management as "a framework and flexible decision-making process for ongoing knowledge acquisition, monitoring, and evaluation leading to continuous improvements in management planning and implementation of a project to achieve specified objectives." A concerted effort has been made to insert science into the planning, policy and management components of the Act.

12.6. The Delta Plan

The goals set out in the Delta Reform Act of 2009 are to be met through the Council's development and adoption of a Delta Plan. The Delta Plan is to lay out policies, recommendations, and management goals for the Delta through 2100. The Delta Plan will be a "living document" with periodic updates and use adaptive management principles to guide planning, implementing and revising the plan. Key components of the Delta Plan require substantial scientific input. The Delta Science Program took the lead on chapters and sections concerning 1) science and adaptive management, 2) ecosystem restoration, and 3) water quality. The dual challenge of providing water security and decreasing threats to aquatic biodiversity are at the core of this new plan.

The Delta Science Program provides scientific input to decision makers charged with adopting the new Delta Plan. The Program takes an ecosystem-based approach to supporting research in the Delta with a commitment to high quality science, communicating science to a diverse audience, promoting ecosystem-based management and adaptive management, and carrying out rigorous evaluation of past and future projects. The Delta Science Program also attempts to provide independent scientific oversight, integrate across program and agency issues and mandates, ensure that decision makers have reliable information concerning complex Delta issues, and play the role of "honest broker" among competing interests. This involves the convening of public workshops to discuss contentious issues, the constituting of independent review panels to openly review scientific documents, and support of targeted science to address key uncertainties affecting policy decisions. Science support for the current planning exercise has particularly focused on linking emerging scientific understanding in the Delta to responsive policies and recommendations for flow objectives, delimiting best available science, adaptive management, ecosystem restoration, and water quality (http://www.deltacouncil.ca.gov/delta-plan).

The Delta Plan is being developed through a transparent and collaborative process (Figure 12.4). When adopted, the plan will have undergone seven public drafts. Following each Council-staff prepared draft, the public was given considerable opportunity to comment on the Delta Plan through meetings of the Council, written comment letters, public workshops and agency stakeholder meetings. To ensure that the best science is used in the development of the

Figure 12.4: *Public meeting of the Delta Stewardship Council. Five of the seven council members are shown*

Delta Plan, the Delta Independent Science Board was asked to provide scientific review on early drafts of the plan. The Council will vote to adopt the final draft of the Delta Plan in the spring of 2013. Once the Delta Plan is adopted and approved as State of California regulation, the Council will have the legal responsibility and authority to implement the plan.

Implementation of the Delta Plan will continue to rely on the use and development of best available science. Current drafts of the Plan commit the Delta Science Program to play a key role in 1) the continued development of science-based performance measures for the Delta Plan, 2) the development of landscape-scale conceptual models for informing restoration decisions in the Delta, 3) synthesis and evaluation along with communication of science to inform adaptive management of the Delta Plan, 4) the coordination of workshops to inform policy decisions related to Delta environmental stressors, and 5) the development of a Delta Science Plan that utilizes an open and collaborative process in developing an institutional and organizational structure for conducting Delta science activities in an efficient, collaborative, and integrative manner.

The Delta and the rivers that flow into the Delta are at a crossroad. The Delta is changing rapidly as human population growth, invasive species introductions, the risk of earthquakes, increasing sea level rise, continued land subsidence, deteriorating water quality, altered hydrodynamics and a changing climate constitute multiple stressors upon the system. Critical questions such as the best way (environmentally and economically) to convey water through (as is currently done), around (involving a canal that diverts water before reaching the Delta), or beneath (large tunnels transporting water underground) the Delta and whether habitat restoration can effectively mitigate for water exports remain unanswered. A new governance structure (the Delta Stewardship Council) with a science program and an independent science board was created by legislation in November of 2009. The Delta Stewardship Council must institute policies and make recommendations to achieve the coequal goals of a more reliable water supply for California and protecting, restoring, and enhancing the Delta ecosystem. As required by law, the Council will use BAS and a science-based adaptive management strategy for decisions on ecosystem restoration and water management. The planning process is actively ongoing with long-term conservation of the Delta ultimately in the balance.

Scientists interested in conserving and restoring rivers must be more capable at working the interfaces between science, policy, planning and management

12.7. Science and policy in conservation of the Murray-Darling Basin

The Murray-Darling Basin (MDB) drains just over a million square kilometers of south-eastern Australia and is divided among five different states (Figure 12.5), each with different systems of water entitlements and management. Some two million people live in the MDB, which supplies most of the water for another million people downstream in South Australia. Earning it the name "Australia's bread basket", the MDB contributes some US$15 billion of agricultural produce each year in Australia, of which US$5.5 billion is derived from irrigation (~45% of national irrigation produce; MDBA 2010). However, scientists have shown that much of the system is in poor health (see Sustainable Audit section later) and many catchments are grossly overallocated (in some cases, over 100% of the entitled water has been allocated for human use). Public and political awareness of the severity of the MDB's plight peaked after a decade-long drought (ending in 2010) on top of some two centuries of unsustainable river exploitation by European settlers. Why did it take so long to react to the environmental damage and how can the ecological resources of the Basin be restored and conserved?

Since Federation in 1901, state governments have squabbled over allocation of water in the system. Not surprisingly, states in the upper reaches have been accused of reducing water resources for downstream states. State governments used water as a tool to promote rural community growth and there was little

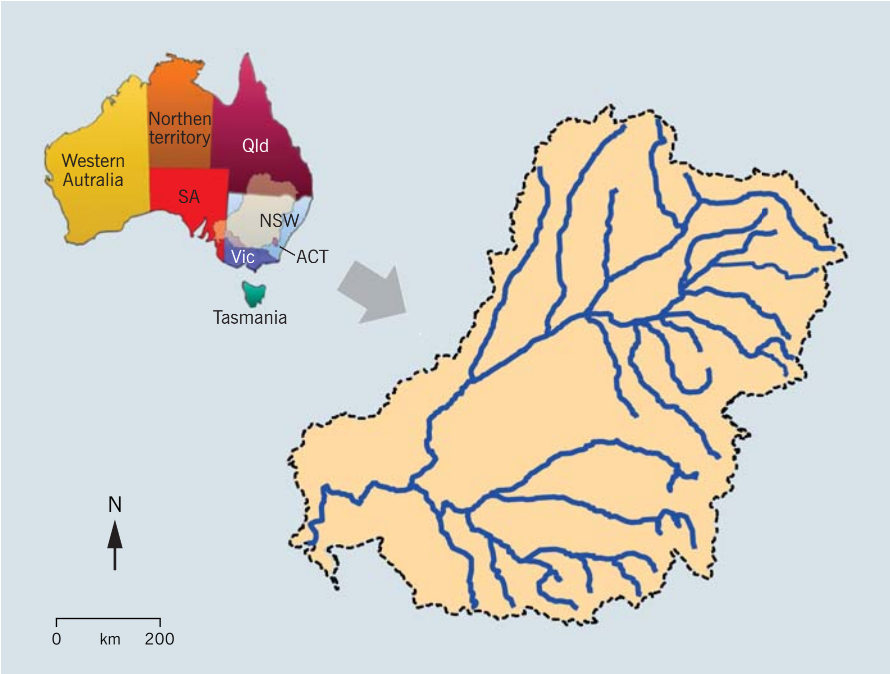

Figure 12.5: *The Murray-Darling Basin (shaded) comprises two main rivers – the Darling River to the north and the Murray River to the south – flowing southwest and draining the states of Queensland (Qld), New South Wales (NSW), the Australian Capital Territory (ACT), Victoria (Vic) and South Australia (SA)*

concern about environmental issues or conservation. Sharp increases in diversions in the second half of the twentieth century intensified competition among water users and caused serious environmental problems. State governments stepped back from promoting irrigation interests and, instead, adjudicated water usage among competing stakeholders. In the meantime, the Commonwealth government sought to manage the basin as a whole, requiring a policy instrument that would coordinate the two levels of government. This spawned the Council of Australian Governments in 1992 whose deliberations led to the National Water Initiative (NWI) in 2004. The NWI, among other actions, sought to cap extractions at 1994 levels. However, when these agreed reforms failed to deliver sustainability, the Water Act 2007 was enacted.

This Act created the Murray-Darling Basin Authority, a body charged with developing and implementing a Basin Plan as an integrated approach to manage the MDB's water resources. Central to the plan is a "sustainable diversion limit" (SDL) set for the whole basin, with diversion limits also set for sub-basins. Best available science was explicitly requested to help set these limits, taking into account environmental demands, water quality and salinity as well as changes in runoff predicted as a result of climate change, bush fires and new agricultural activities. Working within the Basin Plan's policies, States are required to develop water management plans for approval by the Federal Minister but state governments still retain control over their water resources.

Although many scientists had worked on the ecology of the MDB for decades, there was little catchment-wide research and coordination among individual scientists was limited. Often, scientific research was commissioned in a reactive way. For example, when a 1,000-km long blue-green algal (cyanobacteria) bloom threatened farming communities along the Darling river in 1991, research projects on blue-green algae burgeoned. Only when "science champions" such as Professor Peter Cullen (Box 12.3) coordinated efforts within cooperative research centers while also wielding considerable political influence with state and federal governments did the results of scientific research start to effectively guide policy development and river basin conservation. Other champions, such as Professor Richard Kingsford, coordinated groups of scientists to become involved in workshops on rivers of conservation significance (e.g. the Paroo, Chapter 13), prepared consensus views on environmental issues co-signed by fellow scientists (e.g. scientific statements on the Basin Plan, http://www.wetrivers.unsw.edu.au/2012/04/scientific_statement_pbp/) and appeared frequently in the media, promoting the role of good science in river conservation and management.

12.8. The Sustainable Rivers Audit – Murray-Darling Basin

In 2004, the first formal coordinated audit of the rivers of the Murray-Darling Basin (MDB) was carried out, supervised by a panel of four independent ecologists. This program, the Sustainable Rivers Audit (SRA), assessed five ecosystem components: hydrology, physical form, vegetation, macroinvertebrates and fish (methods described fully in Davies et al. 2010). Metrics derived from assessments of these components in 23 rivers of the MDB confirmed the dismal state of the system. Only one river (the Paroo, Chapter 13) was rated in Good Health and only two other systems were deemed in Moderate Health. The rest were assessed to be in Poor or Very Poor Health (Davies et al. 2010).

These data were crucial for scientists, managers and policy makers in their application of the Water Act 2007 to determine sustainable diversion limits for each of the rivers of the MDB and underpinned what is known as the Basin Plan. The Basin Plan is a system-wide attempt to protect and restore the ecological and other values of water-dependent ecosystems of the MDB so that the ecosystems remain healthy in the face of climate change. To achieve this, "long-term average Sustainable Diversion Limits" (SDLs) were derived from the hydrological and ecological data, combining assessments of surface and

Box 12.3

Peter Cullen – shining example of a scientific champion

Figure 12.6: *Professor Peter Cullen, one of Australia's greatest science champions*

Professor Peter Cullen (1943-2008) was an exceptional champion of water reform in Australia. His personal attributes of great humanity, a powerful work ethic, scientific understanding, political awareness, oratory skill and dry humour allowed him to influence Prime Ministers and state Premiers, irrigators and farmers, scientists and journalists. Early on, he acknowledged the "turbulent boundary" between scientists and managers (Cullen 1990) and devoted the next two decades of his life to improving dialogue between two groups whose ideologies, backgrounds and time frames often differed. Although he was a strong advocate of the role of science in water resource management (for example, founding and directing the Cooperative Research Centre for Freshwater Ecology), he once said: "Scientists commonly hold strong values about desirable outcomes, and should be welcome in the political debates as society grapples with the various issues. However, they should not expect their scientific standing gives them any special right to decide value questions for society. Their science needs to inform the debate, not replace the debate" (Cullen 2006). He mentored many scientists in how to become usefully involved in political debates and discussions, urging them to make a more effective contribution in situations where all interests do not necessarily welcome the scientists' messages. Most of all, he constantly argued that scientists have an obligation to ensure that their knowledge and insights are available to the community that funds them.

Peter Cullen successfully bridged the gaps between science, resource management and policy. He saw the "big picture" and not only described the problems but suggested solutions. He was an influential member of the powerful Prime Minister's Science, Engineering and Innovation Council, and he proposed many of the research and policy threads in the Australian National Water Initiative. He won many prestigious international awards (summarized in Lake et al. 2010) for his work in water reform and environmental management. Most importantly, he was a consistent and effective champion for the role of rigorous science in water resource management, policy and the conservation of freshwater biodiversity (e.g. Cullen 2003).

groundwater resources as well as the SRA results. The Act specifically requires the Basin Plan to identify risks to the condition and availability of the MDB's water resources and to identify strategies to manage those risks. A guide to the proposed Basin Plan was released in October 2010 and met with instant strong and demonstrative opposition by irrigators in some quarters (Figure 12.7) who predicted financial ruin.

Even before this release, in response to concerted lobbying by vested interests, the recommendations for the environment's share of the diversions were reduced from an initial estimate of 7,600 GL per year down to 3,000-4,000 GL per year (in October 2010) for public discussion. The guide to the proposed Basin Plan suggested that the full range of natural variability would be encompassed within 3,000 – 7,600 GL per year. However, the strong reaction to the suggested SDLs led to a reassessment by the MDBA. In November 2011, after considerable further consultation, two parliamentary inquiries and resignation of the Chair of the MDBA Board and Chief Executive Officer, the MDBA produced the proposed draft Basin Plan (http://www.mdba.gov.au/draft-basin-plan/draft-basin-plan-for-consultation). The media release (28 November 2011) stated: "More recent and robust modelling has shown that key environmental objectives can be met with a lower volume than the range suggested in the Guide" and so they advocated 2,750 GL per year with a seven-year period (to 2019) to implement this volume. Almost half that volume

Figure 12.7:
Angry irrigators in Griffith, New South-Wales, burn a copy of the "Guide to the Draft Murray-Darling Basin Plan Volume 1" after it was released in October, 2010

had been obtained by water buybacks and improved infrastructure by the end of 2011.

Although there is general agreement that the MDB is in poor ecosystem health, the setting of SDLs upset many irrigators objecting to cutbacks on their water allocations, even though these would be paid for by taxpayers. SDLs are to be achieved by a combination of water buyback and investment in infrastructure, and the Australian government has made a commitment to "bridge the gap" between current levels and proposed levels of water diversions without affecting entitlement or allocation reliability (http://www.mdba.gov.au/draft-basin-plan/draft-basin-plan-for-consultation). By 2019, it is expected that buybacks and infrastructure investments will have achieved the reductions in diversions. To support the plan, the Australian government committed over US$9 billion to the MDB up to 2019.

The Water Act 2007 stipulates that SDLs will reflect an environmentally sustainable level of water removal. Scientific advice underpins determining how much water the ecosystem "needs", when it needs it (i.e. seasonal flow regimes) and how these amounts will differ from year to year in response to climate change and natural annual variability in flows. Socioeconomic studies have also been carried out to ascertain the likely effects of different SDLs on various Basin communities. Results of these studies indicated that the proposed SDLs would not have an unduly harsh impact on some local human communities and, where impacts were likely, what strategies would ease the transition. There was also an assessment of the value of ecosystem services improved by the return of flows, estimated to be some US$3-8 billion (CSIRO 2012). The most recent modelling studies (Young et al. 2011) consider the science to be adequate and argue that 2,800 GL per year would be an appropriate compromise. However, many scientists remain sceptical because they do not believe this amount of water is adequate to fulfil the ecosystem's needs, especially in the face of projected climate change and human water demands in this largely dryland system (Figure 12.8).

12.9. Interacting with managers and policy makers from an Australian perspective

In Australia's Murray-Darling Basin, scientists, managers and policy makers are grappling to find a new way to interact. After two decades of "engagement" under various natural resource programs, the Millennium Drought or "big dry" (*sensu* Prowse and Brook 2011) highlighted that the country's river management plans, policies and best available science had not coalesced to prevent

Figure 12.8:
Much of the Murray-Darling system, such as this section of the Darling River, flows through arid and semiarid country. The water is typically turbid and natural water levels can vary greatly between long periods of low or zero flow alternated by irregular huge floods

massive biodiversity loss. Large tracts of red gums were killed throughout the basin by lack of water, and internationally important wetlands were parched without some natural flows so that aquatic groups such as fish, water plants and waterbirds declined sharply in abundance (Kingsford et al. 2011). Water Sharing Plans were suspended in many NSW rivers when the rules of allocation to water users failed during the long drought (National Water Commission 2009). The plans captured volumes and flood frequency, but did not set maximum limits for the inter-flood interval, the critical dry period between floods.

With opportunities for reform being forged by the Basin Plan and the establishment of the Commonwealth Environmental Water Holder (CEWH) with considerable funding for the buyback of water (US$3.1 billion), the playing field for interaction with scientists is changing. Unfortunately, this is largely driven by who controls science funding. Monitoring of environmental flows is no longer the bastion of state government agencies, with the CEWH contracting scientists to report on outcomes of its environmental water releases. At the same time, government budget cuts in NSW are reducing the state's capacity to meet monitoring and research obligations for rivers. The CEWH has released a framework for monitoring, providing one avenue for debate and coordination. Senior managers from the Murray-Darling Basin Authority (MDBA) and National Water Commission are pushing for greater collaboration among scientists to address complex problems but without a clear investment. However, the freshwater research direction at the Commonwealth (national government) level is uncoordinated and criticised by some as lacking leadership. In response, scientists are organising into clusters to research ecological responses to environmental flows and to tender for monitoring contracts on environmental flows. The days of individual scientists broaching ad hoc research projects with managers and planners have passed.

There is consensus that scientists need to be organised at a broader scale to interact with managers and policy makers, but the nature of this coordination is unclear. Should there be a Commonwealth scientific body to debate and drive collaborative direction? In the past, Land and Water Australia provided a focus for ideas, but the funding programs fostered competition rather than collaboration at a broad scale. Land and Water Australia was abolished late in 2009, leaving a vacuum in science funding. Science and Technology Australia provides an advocacy role for science, but has not stepped into directing or coordinating roles. Could market forces drive scientific consensus if the MDBA and CEWH, for example, set the agenda for broad scale research on multi-stressor problems and demanded collaboration? Ideally, if planning at a basin scale followed a Strategic Adaptive Management framework (Kingsford et al. 2011), interactions between science, management and policy would promote debate and coordinated solutions to the water crisis in the Murray-Darling Basin. A real engagement process (see examples in Chapter 13) could transform the MDBA from an agency "that everyone hates" to an agency that is admired for its leadership by all sectors of society.

12.10. Communicating the role of multiple stressors on the MDB system

Without dissent, a policy commitment exceeding US$9 billion to "save the MDB" was announced before the Australian federal election in 2007 although no-one has ever explained where this monetary figure came from. It had bipartisan support at the federal level. Byron (2011) asserts that if anyone had asked in the Commonwealth Parliament in 2007 "Who wants to save the Murray-Darling Basin", all hands would have risen. However, if the question had been "Does anyone understand the nature of the problems facing the MDB, how we got into this mess, the options for getting out of the mess, how long that will take, how much it will cost, and whose cooperation do we need to succeed", then Byron suggests that no hand would have been raised. This illustrates that either scientists have done a poor job of communicating the impact of multiple stressors on the MDB or that politicians and managers have been poor listeners – or both.

In an effort to simplify the message for rapid communication, the widely held perception of the "problem" in the MDB is that rivers, wetlands and floodplains are under severe stress and that the cause is excessive extraction of water from the rivers for irrigation. The "solution" is seen in equally simplistic terms: if irrigation extractions are reduced to SDLs and the saved water is re-assigned for environmental purposes, the basin will be restored to a healthy, sustainable system. The pervasive notion is that "all the environment needs is more water" with one corollary seeming to be that "the more water added, the better the environmental outcomes will be". This notion underlies much of the debate about the Basin Plan and because it is half-right, it is hard to refute. Nonetheless, the debate remains critically important and when the number defining how much water will be returned is finally settled, the focus will move to other stressors on the rivers.

More scientists need to be both willing to engage in and be better trained at effective communication with planners, policy makers, decision makers, and politicians

However, environmental decline in the MDB has occurred because of more than just declines in water volumes. There have been changes in the flow regime (seasonal timing and variability of river flows and floodplain inundation), water quality has deteriorated, exotic species (e.g. carp, willows) have invaded the rivers' channels and riparian zones, structures have interrupted flows on floodplains, numerous dams and weirs interrupt the longitudinal dispersal of riverine fauna and flora, and sediment regimes have been altered by inappropriate catchment clearance and land use. Multiple, interacting stressors impact upon the rivers of the MDB and it is impossible to point to a single stressor and claim that it is the main problem.

Scientists have described the multiple stressors repeatedly in unpublished reports, peer-reviewed literature and the popular media. Local governments have spent thousands of dollars attempting to control particular stressors, such as by restoring riparian zone vegetation and decommissioning weirs and dams. Efforts to restore the timing of natural flows and inundation patterns have had some success. However, it is striking that much of the focus of the Basin Plan has been on the SDLs whereas the MDB is afflicted by multiple stressors, some of which are unlikely to be resolved by simply adding more water back into the system. Effective conservation and management of the MDB needs better communication from scientists about the effects and interactions among multiple stressors and how best to ameliorate their collective impacts.

12.11. The need for *champions* for improving the role of science in river conservation

Poff et al. (2003) discussed the need for improving the science used for setting flow criteria in river ecosystems. The highly contentious process of determining flow requirements for rivers to achieve desirable ecological outcomes while ensuring reliable water supplies requires new and emerging science. Scientists need to be viewed and accepted as partners at the table with resource managers and other stakeholders in a collaborative process in managing river ecosystems. This way, scientific understanding, management strategies, and societal goals are effectively integrated. Four recommended steps for strengthening the role of science in managing rivers to meet human and ecosystem needs are: 1) large-scale experiments on existing and planned water management projects, 2) collaborative processes involving scientists, managers, and other stakeholders, 3) integration of case-specific knowledge into broader scientific understanding, and 4) forging new and innovative funding partnerships (Poff et al. 2003).

Ultimately, improving the role of science in river conservation requires scientific champions. Peter Cullen epitomizes such a champion in Australia, and he made major contributions to science, policy, planning and management of Australian rivers (see Box 12.3 and Figure 12.6). The success of a scientific champion hinges on having respect and credibility across the entire sector from fellow scientists to the general public, excellent communication skills, and a work ethic that combines sustained effort and persistence with patience and the capacity to be willing to repeatedly contribute to debates at all levels, even the publicly unpopular ones. These traits are rare in any individual, let alone a trained scientist.

Scientific champions typically achieve more when they work as a professional collective. This is because they can draw on a greater range of skills and expertise, and are likely to have a higher public profile. In Australia, Peter Cullen was a founding member of one such collective in 2002. This independent group, calling themselves "The Wentworth Group of Concerned Scientists" (www.wentworthgroup.org), inserted science effectively into conservation and water resource management in Australia through some highly publicized media releases. The Wentworth Group included leading Australian scientists, economists and business leaders with conservation interests. They produced a series of "blueprints" – readable, closely-argued and brief documents that outlined the environmental problems facing Australia's water resources and explained the causes. These blueprints also presented solutions that would protect river health and Australians' rights to clean usable water, establish nationally consistent water entitlement and trading systems, and engage local communities to ensure a fair transition.

In 2010, the Wentworth Group produced a blueprint on sustainable MDB diversions (http://www.wentworthgroup.org/uploads/Sustainable%20Diversions%20in%20the%20Murray-Darling%20Basin.pdf). This blueprint drew on the best available science to identify the maximum quantity of water that could be taken from the basin and from the 18 sub-basins, assessing the most cost-effective way to obtain the water while assisting local communities to adapt to the changes in water resources. They concluded that the basin's rivers required two-thirds of their natural flow to be healthy and recommended that the environment's share of the diversions should be 4,400 GL per year (from an estimated average annual end-of-system flow of 12,233 GL per year before European exploitation).

The Wentworth Group's success arose from several factors (Cullen 2006). First, their blueprints used clear and simple language and avoided qualifiers and citations of scientific references. Second, they clearly articulated the problems and linked these to realistic, effective solutions. Third, the key messages remained focused and the group shared a vision to pool their expertise to develop integrated solutions to problems. Fourth, the group was not self-interested or simply calling for more research funding. Fifth, the members of the group were well-recognized in their areas of expertise and had substantial media standing and skills. Finally, the group never claimed that the proposed solutions were the only ones or even the best ones, but they suggested the solutions were effective and invited anyone with better solutions to bring them forward. By writing succinct blueprints instead of detailed treatises, by using media in a timely and skilled way, and by being willing to debate their blueprints widely, the Wentworth Group was extremely successful in inserting science into several complex management and conservation debates in Australia. Champions like Peter Cullen and his colleagues have done much for river conservation and restoration in Australia. Similar champions are currently playing critical roles in river conservation worldwide.

12.12. Challenges for inserting science into river conservation

When science is incorporated into planning, policy and management, decisions can also have a large impact on conservation efforts. Inserting science at the outset of planning for river conservation provides policy makers, planners and decision makers with a better understanding of the need for science rather than seeing science as obstructive or slowing the planning and decision-making process. It also is important to communicate the relevance of science and engineering in decision-making to those making river conservation decisions. This may entail repeated, positive and non-confrontational exposure to relevant science (e.g. the Delta Lead Scientist makes regular presentations of relevant and leading scientific papers and findings to policy makers on the Delta Stewardship Council at monthly public meetings). When planners and policy makers see the need for science upfront, they are more likely to seek scientific input. However, demonstrating that science, especially ecological science, provides added value to policy, planning and management decisions can be a challenge for scientists. Ecology concerns itself with relationships between living organisms and their environment, and river ecosystems link climate, hydrology, chemistry, and ecology in ways that can guide good policy and decision-making. Communicating these interactions with good timing and clarity is necessary to the incorporation of current scientific understanding into river management. Science that successfully pushes policy, planning and management forward will acknowledge multiple stressors, point towards well-ordered and manageable steps toward improvement, and provides time points to celebrate situations when science has helped successful river conservation efforts (see Chapter 13).

A key challenge of inserting science into river conservation is access to decision makers and politicians. Sometimes, enabling legislation facilitates scientific input into river conservation and restoration (e.g. the Australian Water Act 2007 and the Delta Reform Act of 2009). The challenge then becomes one of utilizing this access effectively by communicating science in a clear and applicable manner. In other cases, pressure from scientists themselves and the public is necessary to bring scientific information into the decision making process. Democracies have more effectively inserted science into the policy arena with the more open and public institutions that allow due consideration of scientific information. Scientists, however, must realize that the opportunity for input on issues of river conservation and restoration does not guarantee a positive outcome. Decision makers, however, also need to acknowledge that scientific input and application of BAS does not mean repeated solicitation of technical input until the content of that input is finally deemed acceptable. Persistent scientific champions with good communication skills and a broad and interdisciplinary understanding of river ecosystems are most effective in inserting science into policy, planning and management, but science still needs to inform the debate but not replace the debate, as Peter Cullen perceptively pointed out.

12.13. Tactics for enhancing communication and resolving conflict

River conservation typically leads to conflict because when water resources are allocated back to the environment, other users are denied water that could generate income. Multiple and competing values for water at a time when human populations are increasing, water resources are dwindling, water quality is deteriorating, and climate is changing is an inevitable result. In an ideal world, collaborative approaches that allow all stakeholders to express their concerns and feel satisfied with the resolution would predominate. There are many models proposed to promote this collaboration (e.g. Daniell 2011) and some examples where consensus has been achieved, leading to examples with varying success in river conservation (Chapter 13).

More commonly, conflicts arise. These are usually exacerbated by the centralized technocratic management of river basins and water resources coupled with minimal levels of interactive engagement with stakeholders. They arise because stakeholders have different values. The political process provides the forum for contesting these sets of values, and judgments are often made on the basis of short-term popularity rather than long-term benefit (Cullen 2006). Political conflicts are resolved by bargaining and negotiation, aiming to find a solution that will be supported by a coalition of interest groups; a marked contrast to the way that scientists resolve conflicting hypotheses in their research. And yet publication of science can fundamentally influence the political process by providing new information on the condition of resources.

Most conflicts have five key elements (Box 12.4). These elements commonly occur in environmental conflicts but are seldom clearly recognized by the players, hampering conflict resolution or problem identification. Further, some parties in many environmental conflicts are unaware of the tactics used by various interest groups to complicate the issues in an effort to maintain the status quo. River scientists, in particular, seem to be unaware of these tactics which range from repeated denial of the problem and the engagement of advocacy organizations to confuse issues further through to attempts to silence scientists who work in government agencies on the grounds that they should not be involved with policy (Cullen 2006).

Few aquatic scientists receive formal training in conflict resolution. We suggest that in addition to improving techniques of scientific communication with stakeholders in conservation debates, approaches to conflict resolution that promote joint benefits ("negotiation theory") should be taught to aquatic scientists entering political and management debates. These approaches would include adoption of problem-solving behaviour, minimizing "contentious behaviour" and understanding pro-social motivation where compromises are perceived as foregone gain rather than overall loss (e.g. Gelfland and Brett 2004). A few such courses in conservation conflict resolution exist (e.g. Society for Conservation Biology, Smithsonian National Zoological Park), although these appear to target terrestrial rather than aquatic scientists.

Box 12.4

Conflict resolution in river conservation: Scientists and elements of environmental conflict

Multiple and competing demands for water in most rivers lead to conflict. Scientists can play an important role in helping resolve environmental conflicts. These include: identifying the problem's scope and implications, helping develop and evaluate strategies to solve the problem, modelling scenarios with and without a conservation intervention to help illustrate the consequences of particular actions, and monitoring the responses to conservation actions to inform Strategic Adaptive Management. Scientists also can contribute to getting an issue onto the political agenda, especially because they are likely to be among the first to recognize early warning signs of environmental decline (Likens 2010).

However, scientists are seldom trained in conflict resolution. They also must acknowledge that despite their important contributions listed above, conflict resolution is likely to be driven by value judgments and political consensus as a series of trade-offs. To appreciate this, we need to understand the five elements of an environmental conflict (Cullen 2006). These are:

1. Interests, relating to the personal benefit (e.g. financial reward, access to a resource) gained by an individual or group from a particular outcome;

2. Values, relating to personal attitudes to issues such as development versus conservation, social justice, human rights, etc.;

3. Data, including the conflicting parties' trust in the reliability of available information, its relevance to the particular issue and the way the data are used to address the conflict;

4. Structural issues, arising from the boundaries between organizations with different objectives (e.g. environmental protection agencies versus regional development agencies); and,

5. Risks, and the extent to which different parties in the conflict are willing to risk certain outcomes.

These five elements typify efforts to conserve and restore rivers. Their resolution is complicated because interests and values change over time, often in response to changes in economic situation or options for land use. Increasing population densities and predicted climate change are likely to lead to more intensive conflicts. River ecosystems are notoriously unpredictable and responses to interventions are seldom linear and consistent. This complicates the way that data can be used and may also influence judgements of risk. Finally, political will can be fickle and changeable in many countries, influencing the governance and structures of agencies and their emphases. Effective conservation and restoration of rivers rely on more than physical management; institutional management is just as important yet less widely appreciated (Chapter 13).

12.14. Conclusions

Until we better understand how to insert science into policy and planning in river conservation and restoration, much relevant, good science will continue to be overlooked or ignored. Sometimes, this will be intentional, entailing selective "science-picking" or "combat-science" between duelling hired consultants, and may damage the overall credibility of science, limiting its use in subsequent planning. Scientists concerned with river conservation and restoration need to become better trained and experienced at functioning at the interface between science, policy, planning and management. Our experiences in the California Delta and the Murray-Darling Basin provide some guidance on working at these interfaces where river conservation and restoration are major goals and objectives. Some lessons learned include: 1) developing, nurturing and sustaining communication links with policy makers and decision makers, 2) engaging directly in the planning process for major basin-wide initiatives, 3) identifying and supporting science champions for improving the role of science in river conservation, and 4) learning tactics for enhancing communication and resolving conflict. Successful river conservation and restoration will require more scientists willing to engage in planning, policy and management and better preparation for these scientists to work effectively in these allied fields critical for sustaining healthy river ecosystems.

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Chapter 13

Good News: Progress in Successful River Conservation and Restoration

Andrew Boulton, Cliff Dahm, Lindsay Correa, Richard Kingsford, Kim Jenkins, Junjiro Negishi, Futoshi Nakamura, Peter Wijsman, Fran Sheldon and Peter Goodwin

Worldwide, examples of successful river conservation range from almost complete protection (e.g. Paroo River, Australia) to substantial large-scale restoration of channel form and flow regime (e.g. Kissimmee River, USA). Applying a framework of Strategic Adaptive Management across these examples will help us more consistently succeed in river conservation and restoration.

13.1. Successful river restoration

What is successful river conservation and restoration? In this chapter, "successful" is defined as more than improvements in biodiversity (Chapter 1) and in ecological criteria (e.g. for ecologically successful river restoration, Palmer et al. 2005); we also include improvements in social, economic and political values of rivers. These latter three values encompass protection of aesthetic, natural and functional economic aspects (i.e. ecosystem goods and services, Chapter 1) of rivers. Success in attaining these social values is underpinned by political resolution of the tension between solely economic development of river systems versus the protection and conservation of their natural values. Although ecological science and communication have crucial roles to play in the resolution of this conflict (Chapter 12), successful river conservation and restoration also requires robust institutions and effective political governance, often across borders.

Successful river conservation means improving social, political and economic values of rivers as well as ecological aspects. It is a broad and complex task

Further, we define "conservation" more broadly than simply protecting species diversity in an area. In this chapter, we regard conservation to include activities such as active restoration, removal or mitigation of threats, and active management. Successful conservation relies on effective management, supported by well-designed monitoring and evaluation programs with clear goals and an underlying model of how the conservation actions are intended to benefit the ecosystem, increase biodiversity, and enhance resilience (Chapter 11). For true success, there must be explicit links with learning from the conservation strategies and their management, assessing how these can be improved and generalised to other rivers. This is the central theme of our chapter.

In this chapter, we outline a framework for considering the spectrum of river conservation needs and approaches – ecological and sociological – that matches the extent of anthropogenic development of different rivers. This framework is presented at the scale of the entire catchment but acknowledges that most conservation and restoration efforts occur at the local scale, with varying catchment-scale benefits. We present six case studies of successful conservation across the world. These studies focus on: 1) setting and defining the desired future condition and goals for conservation, 2) identifying management options, 3) planning and implementing one or more strategies to conserve each river, considering the resources available and the spatial and temporal scales of the conservation efforts, and 4) evaluating and learning from the process. We conclude by reviewing the challenges to improving the success of future conservation of rivers and their catchments.

13.2. Using Strategic Adaptive Management to successfully conserve rivers

River ecosystems and human livelihoods are tightly linked and complex social-ecological systems. They must be managed together. Chapters in this book so far have described the many threats to river biodiversity conservation, painting a gloomy prognosis for most of the world's rivers. To address these problems, the insertion of rigorous and timely science in effective conservation has been emphasised. However, adoption of scientific information must be balanced with adopting social, economic and political values in a strategic approach. This approach needs to formalise, institutionalise and operationalise adaptive management across integrated natural and human systems that operate at large spatial scales (e.g. multiple adjacent drainage basins) and that persist for long periods of time (e.g. decades to centuries). Although there is no panacea for conserving all aquatic ecosystems, Strategic Adaptive Management (SAM) is a management framework that has great potential because of its interlinked processes for navigating complexity and learning (Kingsford et al. 2011; Kingsford and Biggs 2012).

Adaptive management acknowledges the inherent uncertainties of dynamic and unpredictable ecosystems such as rivers but tests these uncertainties through progressively improving management. After nearly three decades of adaptive management promoting scientific experimentation as the central strategy, emphasis is changing to promote a strategic approach that focuses more on the adaptive *integration* of science into social, economic and governance processes. Managers, rather than scientists, play the central role. The key is the progressive value-laden identification of goals and objectives through a hierarchy, leading to scientific understanding. This quantification of systems and measurement of indicators stimulates action when thresholds of potential concern are exceeded or when targets for rehabilitation are required (Kingsford and Biggs 2012).

Broadly, SAM follows four steps. The first is setting the desired future condition (Box 13.1), informed by the context of STEEP (Social, Technological, Economic, Environmental and Political) values and feedback from the subsequent steps (Figure 13.1). The second step identifies the management options, predicting outcomes, testing their acceptability and selecting an option or combination. In the third operational step, we plan and then implement the management option(s) and measure and monitor the identified indicators, ensuring the human and financial resources are available to achieve these objectives. The final step, evaluation, is an iterative learning process that feeds back into the other three steps (Figure 13.1). After intervention (e.g. environmental allocation of water), indicator data are analysed to assess the intervention's effectiveness in progress towards the desired ecological condition. This may include adjustment of the models or objectives, a process that must be communicated to all stakeholders for learning.

Application of SAM to rivers in South Africa and Australia (case studies in Kingsford et al. 2011) has shown promising results but is severely challenged by the complexity of river ecosystems, the size of their drainage basins and overlapping governance complexity. However, the framework is valuable because it integrates across institutions, promotes co-learning, provides explicit decision-making and increases the confidence and morale of managers. Most importantly, SAM can incorporate the intractable and complex social and ecological dimensions that have often led to management failure in previous efforts at river conservation. It also provides a way of linking science explicitly to management.

Box 13.1

Setting goals for a "moving target"

Restoration ecologists agree that all conservation and restoration strategies must have a clear target or "guiding image" (e.g. Palmer et al. 2005). In SAM, this guiding image is termed the **desired future condition**, a "moving target" because ecological systems are constantly changing, often unpredictably. Consequently, setting this target means setting a series of interim targets and refining these over time in response to changes in the ecosystem. As the desired future condition is likely to negatively affect water access by some stakeholders, setting these targets must include effective engagement to establish institutional, cooperative and governance processes (Figure 13.1).

The desired future condition must include **an explicit vision** of the expected endpoint, the **vital attributes** of the endpoint (to focus planning) and the factors that constrain or threaten these attributes at multiple scales. It also needs to incorporate **a hierarchy of measurable objectives** where higher-order objectives capture intent and lower-order ones link to "on-the-ground" interpretations. For example, a lower-order objective may be to fence off riparian zones from cattle-grazing to fulfill the higher-order objectives of promoting recovery of riparian vegetation from the seedbank, reducing erosion and compaction from cattle access, and reducing nutrient inputs from cattle excretion. Finally, setting the desired future condition entails agreement on **a set of key thresholds/targets and indicators** that can be measured adequately to demonstrate progress towards the target (Kingsford and Biggs 2012).

SAM can be applied to the conservation and management of all rivers, across the disturbance spectrum from almost pristine systems through to rivers that are heavily exploited for human needs or that flow through heavily urbanised areas. Depending on the desired ecological condition (step 1), management options for conservation (step 2) and their operation (step 3) can draw from a range of physical and institutional management actions (Table 13.1, Pittock and Finlayson 2011). Physical management actions are active changes "on the ground" (e.g. controlling invasive species, recovering more natural flow regimes) that seek to restore fundamental components of the river ecosystem's biodiversity, integrity and function. Institutional management actions (e.g. policy development, education and training, financial management) aim to improve governance and legislative processes and focus on social, economic and political aspects.

The relative demand for each form of management action varies according to the degree to which the ecosystem is impacted. For example, management of a river system with minor flow disturbance may focus on other threats (e.g. invasive species) and only need limited institutional management (e.g. land use planning, monitoring and research, flow protection, etc.) whereas a seriously

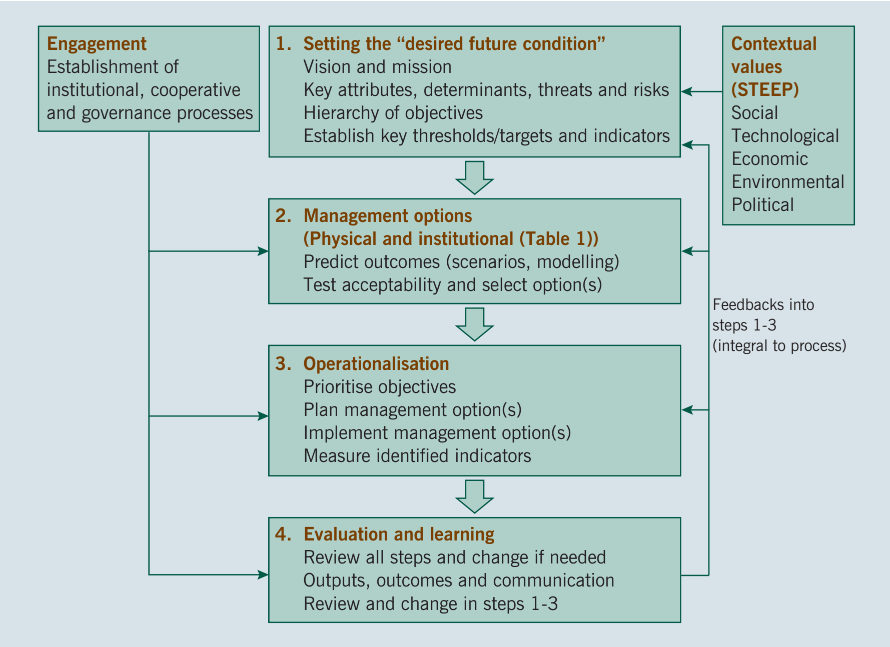


Figure 13.1: *The four steps in the Strategic Adaptive Management (SAM) framework*

Source: Modified from Kingsford et al. (2011) and Kingsford and Biggs (2012).

impacted system may require a full suite of physical and institutional management actions, supported by effective and clear policies (Pittock and Finlayson 2011). Table 13.1 illustrates the spectrum of the varying degrees to which these approaches are or may be used in the case studies that follow.

13.3. Work in progress: Six success stories

Below, we present six case studies ("works in progress") from around the world (Figure 13.2) as examples of successful river conservation or restoration. These span the spectrum from protecting areas that have had little human impact through to severely degraded rivers that need active management and restoration. Each example has elements of SAM and varies in its need for physical or institutional management (Table 13.1).

13.3.1. Murray-Darling Basin's last free-flowing river: The Paroo River, Australia

The Paroo River is a northern tributary of the Murray-Darling Basin (Figure 13.2) and drains a semi-arid catchment of 73,600 km^2, from the state of Queensland

Table 13.1: *Actual or potentially useful physical and institutional management actions (from Pittock and Finlayson 2011) applied to six case studies of successful river conservation. Relatively unimpacted rivers (e.g. Paroo) may only need a few institutional management actions to continue to protect them whereas heavily altered rivers (e.g. Lower Rhine, Cheong Gye Cheon) will require a fuller suite of physical and institutional management actions. X = actions already done, I = intended actions*

Actual or potentially useful action	Paroo River (near-natural)	Kissimmee River (restoration)	Napa River (restoration)	Kushiro River (restoration)	Lower Rhine (restoration)	Cheong Gye Cheon (urban)
Physical management						
Recover flow regimes		I			X	
Reconfigure channels, floodplains and/or associated wetlands		X	X	X	X	X
Improve water quality (e.g. reduce pollutants and nutrients)		X	X	I	X	X
Conserve natural vegetation (including riparian zones)		X		X	I	
Control excessive erosion		X	X	X	X	X
Recover lost surface water-groundwater linkages		X		I		
Nurture and maintain "protected areas"	X	X		X	X	
Adopt native species recovery programs				X	X	
Removal or mitigation of in-stream barriers to dispersal		X	X		X	X
Flood control (to protect assets and restore river integrity)		X	X	X	X	X
Restore in-stream and riparian habitats		X	X	X	X	X
Institutional management						
Research, monitoring and assessment		X	X	X	X	X
Management institutions (e.g. support and guidance by government agencies, local community)		X	X	X	X	X
Integrated river-basin management		X	X	X	X	
Financing for management and water buy-backs		X	X			X
Legal and legislative protection (e.g. Ramsar, national parks)	X			X		

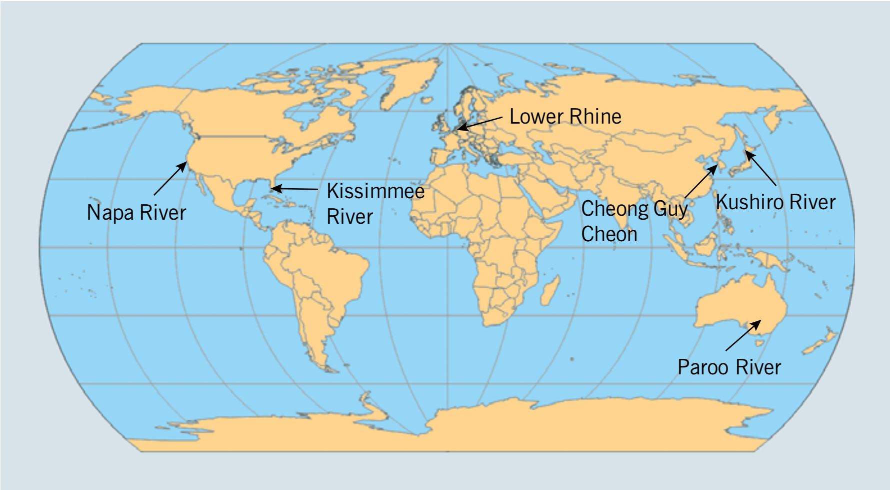


Figure 13.2: *Locations of case studies of successful river conservation and restoration*

Source: Free World Maps: http://www.freeworldmaps.net/.

to New South Wales. Like many dryland rivers, it has a highly variable flow regime resulting in a "boom-and-bust" ecology, typified by brief but spectacular "boom" periods of rapid proliferation of plants and wildlife during floods that are then followed by long "bust" periods when all but a few crucial refugial wetlands dry out. Wetlands of the Paroo such as the Currawinya Lakes can support more than 280,000 birds of over 40 different species, including many breeding species such as the Australian pelican *Pelecanus conspicillatus* (Figure 13.3). At times, the lakes may sustain more than half the world's population of freckled duck *Stictonetta naevosa* (Kingsford and Porter 1994). The river has high conservation significance; there are two wetlands of international significance listed under the Ramsar Convention (Currawinya Lakes and the Paroo River wetlands, including Nocoleche Nature Reserve) in the mid part of the river, and the Paroo-Darling National Park contains the Paroo River overflow lakes.

There were early applications to divert water from the Paroo River for irrigation (Kingsford 1999), despite the likely devastating effects on this river ecosystem. The problem was exacerbated by political polarization across the borders of the States spanned by the Paroo River (Kingsford et al. 1998). A period of considerable argument followed within and outside government about the future policies for the river, primarily triggered by increasing interest in water resource development. Local landholders, dependent on natural (non-irrigation) flows for their grazing income, and scientists drove policy for the river towards protection. In 2003, the New South Wales and Queensland governments agreed to

Figure 13.3: *Australian pelicans take flight during an aerial survey of waterbirds of the Paroo River. Inundated riverine wetlands like these are crucial oases for wildlife that take advantage of the occasional "boom" periods when flooding occurs*

protect the flows in this river (and protect shallow alluvial groundwater) from extraction through an intergovernmental agreement, and in 2007 the wetlands were Ramsar-listed as wetlands of international significance.

Although not enshrined in legislation, the agreement influences water management planning in the two states and still has widespread support. Unfortunately, there is no national framework for the protection of free-flowing rivers in Australia and so the Paroo River remains vulnerable to changes in state policies. Despite this, the wetlands' status as Ramsar sites requires that any future development on the river is subject to assessment under the Commonwealth Government's Environment Protection and Biodiversity Act 1999, as a matter of national environmental significance.

The prognosis for the river remains good, given the considerable discussion and agreement developed to protect the river. There is also considerable opportunity to develop a SAM process for the different protected areas on the Paroo River which would allow a focus on other potential threats to the river and its dependent aquatic ecosystems (e.g. invasive species, tourism). To effect this approach requires commitment by management agencies to develop SAM planning for the key protected areas on the Paroo River. The process could also be scaled up to the entire catchment through intergovernment processes. Nonetheless, even with complete regional protection, the basin remains threatened by global stressors such as climate change and global pollutants (Chapters 1 and 5).

13.3.2. Restoration of channel complexity: The Kissimmee River, Florida

The Kissimmee River is the main tributary of Lake Okeechobee, which feeds the Everglades in southern Florida, United States (Figure 13.2). The Kissimmee River once meandered for 165 km through central Florida, and its floodplain (Figure 13.4), reaching up to 5 km wide, was inundated for long periods of time by heavy seasonal rains from July through December. Native wetland plants, wading birds and fishes thrived in the river and riparian wetlands. Prolonged flooding in the Kissimmee basin in the 1940s led to plans to deepen, straighten and widen the waterway. The Kissimmee River was channelized in the 1960s by cutting and dredging the C-38 Canal, 10 m deep and 100 m wide, straight through the river's meanders (Figure 13.5). Although the project provided flood protection, it also destroyed much of a floodplain-dependent ecosystem that nurtured hundreds of species of native fishes and wetland-dependent birds and animals. More than 90 percent of the waterfowl that once used the wetlands disappeared. After the waterway was channelised, it became depleted in oxygen during the warm months of the year and the fish community changed dramatically.

Figure 13.4:
Up until the 1950s, the Kissimmee River meandered across its floodplain, providing a variety of habitats for a high biodiversity of native birds, fishes and water plants

Figure 13.5: *The C-38 Canal, constructed in the 1960s, slashed through the original floodplain, altering natural patterns of inundation. Remnant meanders now starved of water can be seen in this aerial photograph taken circa 1990*

The Kissimmee River Restoration Project (KRRP) was authorized by the US Congress in the 1992 Water Resources Development Act due to growing concerns about habitat loss and environmental degradation. After extensive planning, restoration began in 1999 with backfilling of 13 km of the C-38 Canal (Figure 13.6). Continuous water flow was re-established to 38 km of the meandering Kissimmee River, and seasonal rains and flows now inundate the floodplain in the restored area. Eventually, the KRRP will return flow to 64 km of the river's historic channel and restore about 12,000 ha of river-floodplain ecosystem. The restoration project – a 50-50 partnership between the South Florida Water Management District and the U.S. Army Corps of Engineers – is projected to be complete by 2015 at a cost of approximately US$980 million. Land acquisition of over 40,000 ha is mostly complete, costing about US$300 million.

Figure 13.6:
This photo, taken on February 9, 2001, shows the back-filled canal flanked by areas of degraded spoil. In the foreground, the remnant river channel has been reconnected across the back-filled canal to link up with an oxbow meander

One key element of the KRRP is a comprehensive ecological evaluation program, matching best practice in SAM. This program assesses achievement of the project goal of ecological integrity, identifies linkages between restoration projects and observed changes, and supports SAM as construction proceeds and after project completion. The comprehensive monitoring and assessment program uses relatively simple conceptual models to predict responses to restoration, the learning component of SAM (Table 13.1). To detect ecosystem changes, data were collected prior to major construction phases to establish a baseline for evaluating future responses. These baseline data are then compared to data collected after construction and re-establishment of pre-channelization hydrologic conditions. Observed changes in the system are compared to predictions described by individual restoration expectations to evaluate whether each expectation has been achieved (steps 3 and 4 in SAM, Figure 13.1). Performance measures to predict ecological changes that are expected to result from the project include changes in hydrology, water quality, and major biological communities such as plants, invertebrates, fish, and birds.

Since completion of the first phase in 2001, there have been increases in dissolved oxygen levels, reductions in floating plant cover within river channels, reductions in accumulated organic-rich sediments on the river bottom, recovery of wetlands, and increased populations of waterfowl, wading birds, bass and sunfishes. Monitoring results suggest that after pre-channelization hydrologic conditions are fully restored in 2014, the primary goal of restored ecological integrity in the Kissimmee River and its floodplain will be successfully attained. Restoration of broadleaf marshes along the restored reach of the Kissimmee River has had mixed results. The restoration of signature broadleaf species like arrowhead *(Sagittaria lancifolia)* and pickerel weed *(Pontederia cordata)* has been variable with some marshes having low percentages of these signature species. Reasons for the limited success of broadleaf marsh restoration along the restored Kissimmee River may include flood-induced mortality, establishment conditions not being met, and invasion by an exotic shrub (Peruvian primrose-willow – *Ludwigia peruviana)* (Toth 2010a, 2010b).

13.3.3. A "living" Napa River restores ecosystems and human communities

The Napa River in central California flows through agricultural and small urban landscapes before entering the San Francisco Bay estuary (Figure 13.2). The basin of the 88.5-km river is famous for its wineries and tourism. However, over a century of altering the Napa River for urban, industrial and agricultural needs transformed the once-meandering river into a straight, constrained and incised river. These alterations harmed the river's "health", degrading water quality and fish and wildlife populations.

Within the City of Napa (population 77,000), the river was squeezed by urban development, with little room to expand during winter storms. As a result, Napa has suffered 22 serious floods over the past 150 years (Figure 13.7, Riley 2011), prompting the federal government to authorize the U.S. Army Corps of Engineers (Corps) to develop a flood-control project. The Corps proposed to channelise the Napa River into a straighter, deeper river through the City of Napa, and asked the local Napa community to pay half the project's cost. The community voted to reject the proposed project in 1976 and again in 1977. After a major flood in 1986, the Corps re-proposed their project, but voters again rejected the project (Viani 2005). What came next was a remarkable demonstration of community cooperation, resulting in a river conservation success story.

Key community leaders and diverse stakeholders banded to form the Community Coalition for Napa Flood Management. This group comprised 400 participants, including members of 40 federal, state and local agencies; local architects and engineers; environmental non-profit organizations; agricultural interest groups; and the local chamber of commerce (Riley 2011). After more than 50 meetings between January 1996 and May 1997, a flood-management plan was

Figure 13.7:
In 1940, the Napa River flooded down Main Street and surrounding streets (Napa, California), causing thousands of dollars of damage to businesses and homes

developed that satisfied all stakeholders (Daily and Ellison 2002). The cornerstone of this plan was a set of “living river” principles that value the vitality of fishes and wildlife, connectivity of the river to its floodplain, and the relationship of people to the river.

Dedicated leadership from community members and agency staff underpinned the development of a cooperative “living river” flood management plan. Among those dedicated leaders was Moira Johnston Block, a local citizen, author and founder of the Friends of Napa River, a non-profit organization responsible for inspiring the “living river” principles. She opened the first public meeting to review the Corps proposal with a simple question, “We are a world class community with our wines, towns and quality of life – why can’t we have a world class project that benefits all parts of our society?” This statement transformed the discussion from a single-objective flood management issue to discussion about what could be achieved at the basin scale. Another leader was Leslie Ferguson of the Regional Water Quality Control Board who was instrumental in opposing the channelization of the river and helped lead the charge for considering a multi-objective flood control project at the basin scale. A third leader was Karen Rippey, a local resident and Friends of Napa River member, who persistently rallied support from public officials and motivated local community participation

(Daily and Ellison 2002). These individuals were the "champions" who inspired the development of goals and objectives for a "living" Napa River System, which became the guiding image for the Napa River Flood Management Plan, satisfying all the contextual values of SAM (STEEP in Figure 13.1).

The guiding image was an innovative engineering and landscape design project, aimed at simultaneously returning life to the river and its community. The design included an attractive waterfront promenade above floodwalls on one side of the river and riverbank terracing on the other side, allowing flood flows to spread horizontally into designated areas. A dry oxbow bypass diverted floods during large storms as well as providing additional wildlife habitat and recreational trails during dry periods. Additional design features included downstream tidal wetland restoration (Figure 13.8) to both provide habitat to native species and to hold large floods, replacement or removal of several bridges, and realignment of the railroad through the city. During construction, old industrial sites would be cleaned up and some commercial and residential structures would be removed or relocated.

On March 3 1998, the "living river" plan was approved by a two-thirds vote by the Napa County citizens, who committed to a 20-year 0.5% sales tax increase to

Figure 13.8: *Restoration of tidal regimes and floodplain access by the Napa River has recovered over 200 ha of wetlands as well as providing crucial flood control during winter storms. The top photograph, taken in 1998, shows how levees blocked the tidal action, constraining the river. The bottom panel, photographed in 2002, shows part of the vast area of wetlands restored by the Napa Valley "living river" project*

pay for the flood control and basin improvements. Dave Dickson, a Napa County employee, was instrumental in developing the funding mechanism required from the local community for the federal flood management project which required demonstrating that the project would benefit the entire Napa community – not just those at risk of flooding. Another agency leader, Anne Riley of the Regional State Water Quality Control Board, introduced a key concept into the group's process: despite the complexity of the issues, planning should be completed in 12 months. Community excitement and political will might have waned if the planning process had been extended longer.

In July 2000, work began to improve 9.6 km of the Napa River and 1.6 km of Napa Creek, including the creation of over 160 ha of emergent marsh and 60 ha of seasonal wetlands. Nine bridges were replaced and nearly 70 homes and 30 commercial buildings were removed as part of the restoration (Riley 2011). With further grants from the California Coastal Conservancy, the city restored 243 ha of former floodplain and tidal marsh that had been leveed off and grazed since the late 1800s (Viani 2005). As a result of the initial restoration, 3,000 properties gained protection from 100-year flood events, flood insurance rates fell significantly and waterfront businesses began to thrive (Daily and Ellison 2002; Riley 2011). Additionally, a five-year fish monitoring program found that the restoration was providing habitat to some 75,000 larval, juvenile and adult fishes of 37 species.

The plan has received several awards and inspired additional restoration efforts in the Napa basin, elsewhere in the United States, and around the world (Daily and Ellison 2002). The well-designed and implemented plan has returned life to the lower Napa River. The "living river" is now supported by functional floodplains, best management practices in the agricultural lands, reductions in contaminant loading to the river, healthy ecosystems that support fishes and birds and, perhaps most importantly, proud local communities. Important lessons have been learned in this example of SAM where social, technological, ecological, economic and political values have been combined to yield a mutually successful outcome.

As this approach to restoration is applied across the Napa River basin and elsewhere, the challenges of meeting the contextual values of all stakeholders continues to require strong leadership and dedication to the living river principles despite the challenges associated with changing faces of agency and community stakeholders during multi-decadal restoration efforts. Also, support for monitoring and evaluation remains a challenge for assessing the hydrological (flood management and water quality) and ecological performance as well as the social benefits associated with the restoration actions and land use planning measures.

13.3.4. River restoration in Japan

The Kushiro River in eastern Hokkaido, Japan (Figure 13.2), drains from Lake Kussharo into the Pacific Ocean, with a lowland stretch of some 20 km through the Kushiro Mire. The mire, originally about 20,000 ha, is a distinctive landscape dominated by sedge fens and raised bogs interspersed with swamps and lakes (Figure 13.9). It harbours many unique species, including the endangered Japanese crane which is designated as a natural monument and attracts tourists from across the world. In the 1960s, the national government led a large-scale drainage project to convert marshy areas for human use by straightening tributaries and the main channel. About 30% of the mire landscape was lost in the upper basin and near residential areas. Eventually, the core of the mire was set

Figure 13.9: *The Kushiro Mire comprises a thick peat layer with a distinctive landscape of sedge fens, raised bogs, swamps and lakes. Unique species such as the endangered Japanese crane* (up) *and an endemic subspecies of the flowering plant* Polonium caeruleum (down) *inhabit the mire, attracting tourists from across the world*

aside as a national park in 1987 following the mire's designation as a Ramsar wetland in 1980. The mire seemed to be saved from further degradation.

However, the lack of a basin conservation strategy led to landscape degradation, largely through excessive input of fine sediment from the upper reaches where channel straightening exacerbated channel incision. Scientific studies revealed that abnormal rates of sedimentation entered the reserve area, gradually dried the land, and altered soil properties. This changed the landscape into one dominated by trees, reducing its wetland ecosystem values (Nakamura et al. 2002). Also of concern was the effect on the wetlands of excess nutrients from point sources in the upper basin (Takamura et al. 2003).

In 2003, dialogue began between governments at various levels, local residents, non-governmental organizations, and academics from various disciplines (including ecology, civil engineering, and hydrology) aimed at reviving the degrading mire landscape as a symbol of cultural and economic integrity in the region. The Kushiro Mire Ecosystem Restoration Project (KMERP) started in 2005 with a goal to restore the mire landscape of the 1980s. KMERP not only emphasized the value of ecosystem conservation of the mire, but also its balance with the local agricultural economy, encouraging regional development. Most important for a successful launch of the project was a shared vision among stakeholders that the mire landscape restoration would require measures at the basin scale (some ten times the mire area). This resulted in involvement of an initially reluctant agricultural sector in communities of the upper basin. The project paid as much attention as possible to the principles of SAM, especially in terms of public involvement, and degradation processes were quantitatively assessed prior to any actions (Nakamura and Ahn 2006).

River restoration was considered critical because the natural flow of rivers is the primary driver of the mire landscape. "Full process-based" restoration was impractical in the short term because a proportion of land with straightened river channels in the upper basin was needed for the regional economy. Therefore, a "partial process-based" restoration approach was implemented. Sediment loads into the mire were reduced through revetment works and the construction of settling ponds in the upper basin. In addition, a 2.4-km stretch of the main channel was re-meandered by reconnecting the remaining former channel and backfilling the straightened section in 2010 (Figure 13.10). Flood levee banks were also removed to promote river-floodplain interactions. This is expected to eventually restore wetland vegetation near the site and to trap sediment that otherwise accumulates in the core mire area downstream.

Ecosystem response to the re-meandering has been monitored for multiple years. Fish abundance and species diversity has increased in the mire and vege-

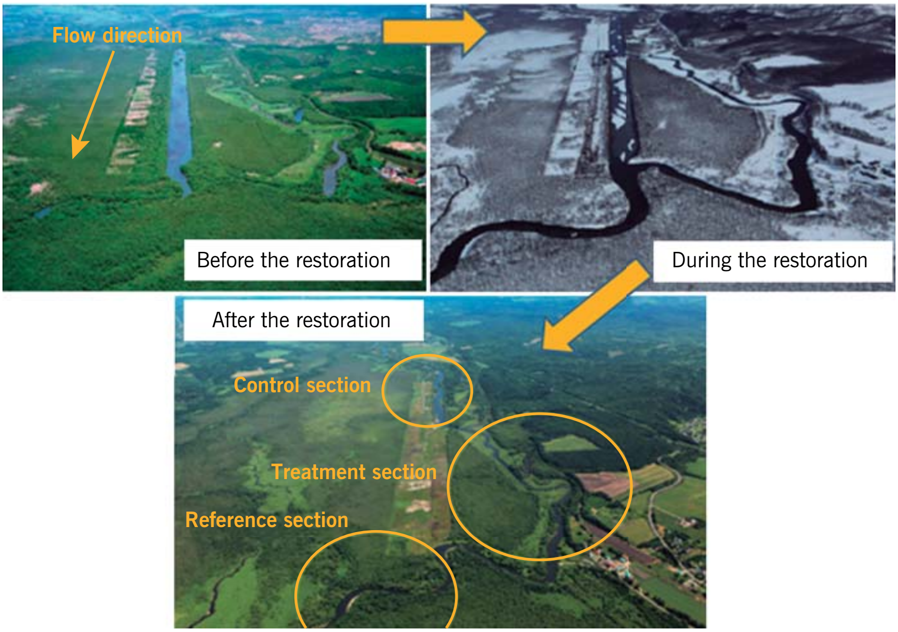


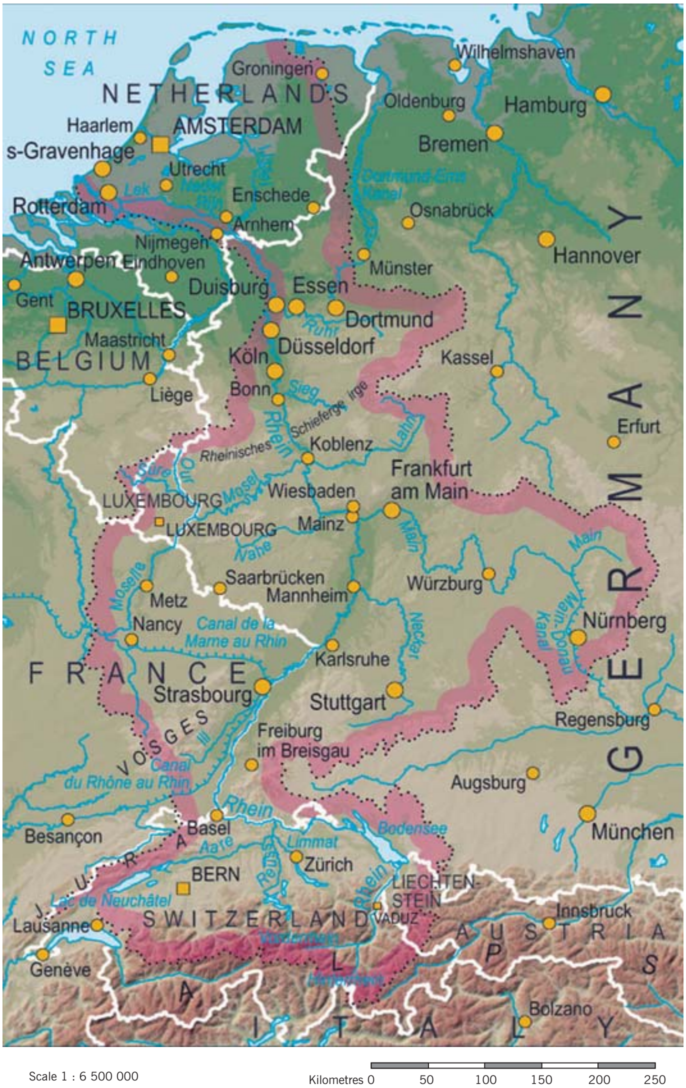
Figure 13.10: *A "partial process-based" restoration approach was implemented in the Kushiro Mire. The top left-hand photograph shows a 2.4-km stretch of the main-channel before it was re-meandered by reconnecting the remaining former channel and backfilling the straightened section* (top right-hand photograph). *Monitoring the effectiveness of the restoration entails comparing the treated section with an unrestored "control" section and a "target" reference section* (lower photograph)

tation characteristics of the mire landscape have partially recovered. Unique to KMERP are programs for local residents to participate in monitoring surveys; local communities benefit intellectually and involvement fosters a stewardship ethic towards the restored mire, matching the learning process advocated in SAM (Figure 13.1).

Restoration of fragile mire ecosystems that may require centuries to develop is made possible by concerted efforts by civil engineers minimizing the impacts of construction. For example, necessary land surface excavation was conducted during winter when land is covered by snow. Channel works were carried out by sequentially dewatering longitudinal channel sections so that heavy machinery caused minimal disturbance in ecologically sensitive riparian zones. Within a year, the landscape in the restored meandering channel resembled that in the reference section. However, it is too early to judge the full ecological success of KMERP because the mire will take decades to recover at the landscape scale. Yet, the launch of a collaborative framework among different stakeholders towards landscape restoration has been a success. This is typically a difficult step in systems with numerous socio-economic constraints where catchments are highly altered, and similar situations abound across an increasingly populated world. The Kushiro Mire case serves as an excellent example of a successful "work in progress" involving channel re-meandering in Asia.

13.3.5. Making Room for the river: Restoration of the Lower Rhine and Rhine Delta

The Rhine basin shares its drainage area of about 185,260 km^2 across nine countries (Switzerland, Italy, Liechtenstein, Austria, Germany, France, Luxembourg, Belgium and the Netherlands, Figure 13.11) with a population of ~58 million people (Uehlinger et al. 2009). The river flows for about 1,250 km with



Source: UNEP, The Global Resource Information Database (GRID): www.grid.unep.ch.

Figure 13.11:
As the Rhine basin's catchment spans nine countries, SAM at a whole-of-basin scale requires substantial coordination and cooperation

an average discharge of ~2,300 m^3/s and services a major economic region. These services include transportation, power generation, industrial production, drinking water for 25 million people, agriculture and tourism. Cioc (2002) characterizes the Rhine River as a "classic multipurpose waterway". Successful SAM and restoration must operate within the constraints of these heavily developed riverine and floodplain ecosystems with their multiple uses, altered hydrology and water quality.

The Rhine basin has a long history of human interaction with the river. Pollution due to domestic and industrial wastewater increased alarmingly after World War II. A significant component to rehabilitating the Rhine has been nutrient and pollution abatement, starting in the 1970s. Improvement in water quality has increased the abundance of the majority of fish species including the return of Atlantic salmon *(Salmo salar)* that was formerly extinct. These fishes are now reproducing naturally in some areas of the basin, fulfilling a flagship role for a charismatic and publicly recognizable indicator of the general progress of improvement of water quality and ecology within the Rhine.

The Lower Rhine flows from Bonn, Germany, to the Dutch-German border (Figure 13.11). About 10 km into the Netherlands, the Rhine diverges into several channels, with water flowing into canals, the Waal River, the Nederrijn River (further downstream called the Lek) and the IJssel River. The surface area of Rhine channels in the Netherlands is ~36,700 ha, including about 28,000 ha of floodplains (Uehlinger et al. 2009). Land use in the Dutch branches of the Rhine floodplains is predominantly grass-production; human-built ecosystems make up about 80% of the floodplains. Water quality (phosphorus, nitrogen and silica) and ecohydrology affecting water-level fluctuations are important factors structuring plankton and plant communities in these floodplain ecosystems (Vanderbrink et al. 1994; Van Geest et al. 2005), and need to be managed for restoring the ecological integrity of this system.

"Room for the River" is an ambitious € 2.3 billion project that is being promoted as both restoration and flood control. It has three primary objectives: 1) improve the overall environmental quality of the Lower Rhine and floodplain, 2) increase discharge capacity for the rivers of the Lower Rhine, and 3) make permanently available extra room to accommodate increased discharge during flood events. Overall, the project is designed to bring greater safety for four million Dutch citizens while improving environmental quality to the lower reaches of the Rhine and the rivers it feeds. Near-catastrophic floods in 1993 and 1995 and the recognition that climate change is likely to increase peak flows in the Lower Rhine have driven the planning effort, providing social, technological and political context for the SAM.

The project involves a range of measures and sub-projects such as lowering floodplains, relocating dikes further inland and lowering groynes (protruding rock-jetties) in the rivers. Thirty-nine locations are targeted for providing more room into which the rivers can flow during times of high discharge. The flood protection measures and environmental quality improvements are scheduled for completion by 2015. The projects are in various stages of implementation with a final goal of increasing maximum discharge capacity of the Lower Rhine through the delta from its current capacity of 15,000 m^3/s up to a peak of 16,000 m^3/s.

One example of the various projects is the depoldering of the Noordwaard (Figure 13.12). A polder is a piece of land in a low-lying area that has been reclaimed from a body of water by building dikes and drainage canals. The Noordwaard polder is influenced both by tidal variations of the sea and discharge levels of the river. The project entails lowering of dikes to create inlets and outlets during times of high water. Parts of the current polder would be under water several times a year, particularly during winter high flow periods. Other parts of the polder would only flood during extreme high discharge periods. Land that is returned to more regular flooding will become floodplain habitat while rarely flooded regions will sustain current land uses (pasture and agriculture). Outcomes include reduced flood risk to the city of Rotterdam and increased floodplain habitat along the river.

Figure 13.12:
The Noordwaard polder project includes "through-flow" areas and green wave-inhibiting dikes that are up to 65 cm lower than the original engineered dikes. Through-flow areas serve to divert some of the higher flows and reduce discharge volumes during winter. This involves a shift in concept from constraining all water in the channel using high levee banks to lowering the levees and allowing flood-water to spread out onto the floodplain but using levee banks to protect houses and infrastructure

13.3.6. Restoring urban rivers: From freeway to waterway in the Cheong Gye Cheon

Most towns and cities started as settlements on the banks of streams and rivers. However, over time, most of these rivers become dammed and channelised, constrained by buildings and industry on the banks, and river health declines from urban runoff containing pollutants. In severe cases, the river becomes an open sewer or an enclosed drain hidden below roadways, car-parks and other impervious surfaces. Restoration of urban streams and rivers is notoriously difficult, largely because only recovering flow regime and structure (e.g. using some of the methods described in earlier sections) seldom resolves the problems of poor water quality and impaired biota. High prices of riparian urban property and the need to substantially alter bankside infrastructure further constrain restoration options and challenge SAM. Nonetheless, public pressure to restore urban waterways is usually intense. Where urban river restoration has occurred, local communities report an improved quality of life, tourism increases and values of surrounding properties rise (Özgüner et al. 2010).

One of the most dramatic river restoration projects of a heavily urbanised area is that of the Cheong Gye Cheon in Seoul, South Korea (Figure 13.2). Once an attractive river (Cheonggyecheon means "clear water stream"), by 1945 the Cheong Gye Cheon had become a silted drain filled with rubbish and contaminated water that offended local residents. The situation was aggravated by the Korean War which left Seoul in a serious crisis as refugees flocked to the city, settling along the banks and further polluting the stream. During the post-war recovery phase, the urban river underwent major transformation from the late 1950s to the early 1970s to cover it over, primarily with a 5.6-km, 16-m wide elevated freeway. This was acclaimed as an example of successful industrialisation and the commercial area burgeoned. However, by the late 1990s, the area was regarded as a source of serious health and environmental problems because of the dense traffic and intensive urbanisation. Carbon monoxide and methane were accelerating the breakdown of the cracking freeway which was considered beyond repair.

In July 2002, the then-mayor of Seoul initiated a project to remove the crumbling freeway and restore the covered section of the Cheong Gye Cheon, now almost completely dry after decades of sedimentation and neglect. Several committees and organisations were established to consider local opinions on the restoration process. The project had immense popular support. However, numerous problems arose during the restoration, including severe engineering difficulties compounded by the deteriorated concrete infrastructure that introduced serious safety issues. However, by late 2005, the "new" Cheong Gye Cheon was opened to the public (Figure 13.13). The water quality issue was

Figure 13.13:
The upper panel shows restoration work in progress on the Cheong Gye Cheon in Seoul (on June 24, 2005). Two years later (lower panel on June 7, 2007), water is flowing where a freeway once passed over the top of the river's course. Vegetation blankets sections of the restored bank and people stroll or sit along the edge of the waterway, once a contaminated drain

addressed by pumping massive volumes of treated water from the Han River and groundwater supplies. Fish species richness rose from 6 to 36 while the number of taxa of insects increased from 15 to 192. The restored stream has reduced local air temperatures and increased relative humidities compared with surrounding city areas (Kim et al. 2009), reversing the usual trends of urbanisation.

The project was expensive (values range from US$281-384 million) and has ongoing and increasing costs to maintain the water supply and sustain the stream. Extensive consultations and conflict-resolution meetings were held throughout the construction period. A detailed environmental monitoring program assessed factors such as air pollution, volatile organic compounds and noise before, during and after the restoration [http://english.sisul.or.kr/grobal/cheonggye/eng/WebContent/index.html]. Although most tourists and urban users consider the project a success, some Korean environmental organisations have criticised the high costs of the project and its limited scope, seeing it instead as purely symbolic and ecologically unsound (Cho 2010). The sides are still lined with concrete and the waterway is monitored for flood control. Further, only a relatively small section of the stream has been restored and the restoration is not ecologically sustainable. Although there is still plenty of scope for application of SAM principles to a broader area of the basin, this spectacular transformation within severe urban constraints has played a key role in changing public attitudes and can be interpreted as having been a successful conservation program in that context.

13.4. Emerging concepts

Worldwide, there are many examples of successful river conservation. We must be inspired by and learn from these, using Strategic Adaptive Management

There are two main themes to emerge from this chapter. The first is that there are numerous examples of successful conservation worldwide. These "success stories" warrant optimism and renewed efforts from stakeholders who seek to enhance ecosystem goods and services provided by protecting or restoring rivers and their adjacent wetlands. However, many restoration projects fail to document recovery and those that do seldom report complete success in all criteria (Berhardt and Palmer 2011). However, we argue that if further loss of biodiversity or degradation in ecological integrity was halted or slowed by a given conservation effort, then that can be deemed "successful".

We agree a common problem is that inadequate documentation or a lack of pre- and post-restoration data prevents assessment of the success and, worse, removes a crucial learning tool (Figure 13.1). When restoration efforts fail but have been properly assessed, managers and scientists can learn from their mistakes and improve future restoration or conservation efforts in light of approaches such as SAM.

The second main theme is the need for integration of rigorous science, community values and action, and effective governance in successful river conservation. This integration needs a framework because the process is seldom effective or efficient without one. We advocate Strategic Adaptive Management (SAM) as one framework for this integration because we believe the emphasis on management rather than science is a sensible direction for change. Of course, rigorous science is still essential. Aspects of this approach have characterised the case studies we present above. However, unless local community members and other "champions" become actively involved in protecting or restoring their rivers, no amount of rigorous science will ensure long-term success. Social, economic and political aspects are as important as ecological criteria to a successful conservation or restoration program. All too often, conservation programs are not limited so much by a lack of knowledge than a lack of public willingness.

Earlier chapters in this book have painted a grim prognosis for rivers. Everywhere, there are deteriorating environmental conditions (Chapters 11, 12), increasing demands for water to support burgeoning human populations (Chapter 1), and many intensifying threats facing the world's rivers (Chapters 2, 3, 6, 7). We urge optimism, initiative and active conservation rather than passive and apathetic resignation to biodiversity loss. Most examples of successful river restoration rely on dedicated people – champions – who refuse to surrender the natural values of rivers in their region and who wish to restore at least part of the natural processes and biota crucial to rivers' ecological integrity and functioning. In our examples of successful restoration and conservation, although projects were planned primarily to benefit the river systems, they also were of benefit to local populations and have been a powerful tool in reshaping public opinion. We hope our examples of varying degrees of successful river conservation and adoption of SAM will help inspire action and indicate strategies that will succeed in other regions.

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13.5.2. Useful links

European Centre for River Restoration: A website that seeks to develop a network of national centres and to disseminate information on river restoration. http://www.ecrr.org/index.html

Global Water Partnership Toolbox: A database of background papers, perspective papers and case studies describing the implementation of better water resource management across the world. http://www.gwptoolbox.org/

Room for the River: Website describing restoration activities on the Lower Rhine http://www.ruimtevoorderivier.nl/meta-navigatie/english/room-for-the-river-programme/

The River Restoration Centre: UK-based advisory website on all aspects of river restoration, conservation and sustainable river management. http://www.therrc.co.uk/rrc_overview.php

Chapter 14

Concluding Remarks on River Conservation

Arturo Elosegi, Sergi Sabater and Andrew Boulton

We have pragmatic and ethical obligations to conserve rivers and their biodiversity. This chapter outlines how and why river conservation is important. To make a difference, we must act as individuals and groups, using water wisely and protecting vulnerable assets such as water quality, riparian zones and aquatic biodiversity.

14.1. Problems and solutions

We take our environment, including our rivers, for granted. Even if we see or hear about the problems of pollution or invasive species in our local rivers, we dismiss the problems as someone else's responsibility. Instead, we focus on our own day-to-day issues while the broader environment around us and our children deteriorates. But we can make a difference and we *can* start to do something *now*.

In this book, we have read about how rivers and their biodiversity face many challenges from human exploitation (Chapter 1), including threats to their hydrology (Chapter 2), structure and architecture (Chapter 3) and water quality (Chapters 4 and 5). We now live in the Anthropocene epoch where human demands for water have severely degraded most rivers and impaired their biodiversity (Chapter 6). This, in turn, has compromised the ecosystem functioning of rivers (Chapter 7), limiting the goods and services on which we rely as a species. These goods and services include the provision of clean water, fish and other aquatic life; sustained aesthetic and recreational pleasure; and the support of healthy catchments, estuaries and groundwaters. Lose these goods and services and we go extinct. There are many pragmatic reasons for conserving rivers.

We have also read about the further problems that beset our rivers. Invasive species occupy many flowing waters and impose severe economic and ecological burdens (Chapter 8). Along the edges of rivers, the riparian zone and its vegetation harbour unique biodiversity and provide crucial ecological functions (Chapter 9) but are vulnerable to pressures on both sides from human uses of the land and the river. At a broader landscape scale, the connectivity within and among river networks and their catchments and marine connections are also disrupted by human activities (Chapter 10).

But there are solutions. There are ways to restore ecosystems to recover the goods and services provided by rivers (Chapter 11). Effective integration of science into planning and policy provide the critical social governance needed in river conservation (Chapter 12), and there are heartening examples of suc-

Figure 14.1: *An oxbow lake in the floodplain of the Colombian Amazon. It hosts a huge biodiversity, including two species of river dolphin, and provides essential ecosystem services to the local societies*

cessful river conservation and restoration (Chapter 13). We must move beyond looking for some-one to blame or waiting for some-one else to fix the problem. Instead, we have the tools and opportunities to start to fix the problem ourselves, and this final chapter explores how we as individuals can all make a difference across a number of levels. We must acknowledge our ethical obligations to protect and preserve species and ecosystems. Conservation is "virtuous" (van Houtan 2006), socially and morally just, and a practical necessity.

14.2. An historical perspective of river use and conservation

It is often said that the past informs the present and the future. By considering how humans have used rivers in the past, we can start to understand why our present attitudes to rivers tend to be so utilitarian; we think of a river as something to exploit and use for economic gain rather than as something we should conserve and protect for its own intrinsic ecological value. Ideally, effective management allows us to do both; conserving rivers while sustainably exploiting goods and services within the systems' capacity to provide them.

Throughout history, humans have acted as thorough transformers of landscapes, forests, rivers, lakes and seas. This action has been gaining momentum (Chapter 1), but the path has been historically consistent since the very onset of the human species.

Conserving rivers is both a requisite for a sustainable future as well as an ethical issue. River heterogeneity and diversity must be passed intact to our heirs

Lets focus this historical perspective on the arid and human-pressed Mediterranean basin which includes most of Southern Europe, Northern Africa, and a small part of west Asia. Most of the Mediterranean basin was densely forested (pine trees, evergreen oaks, cedars) prior to the expansion of human settlements during the Bronze Age. This has been confirmed through pollen records that show significant vegetation disturbance already at 2,000 years before present. The impacts of clearance and the advent of agriculture were not uniform throughout the basin. First, the Cretan expansion and then the establishment of the Greek cities and Rome substantially transformed their immediate landscapes. Wood and metals were raw materials for building houses, weapons and ships, and their search triggered trade and conquest. The Egyptian empire lacked wood and established a large-scale trade with Lebanon (Phoenicia), where they imported the impressive cedar logs necessary in shipyards. In the centuries before the current era, the Tartessos, a people settled in south-western Iberian Peninsula, started mining and smelting metals at an industrial rate in what is called today "Rio Tinto". It was the sheer magnitude of human actions that gave the river its red-wine color, resulting from toxic concentrations of heavy metals. The effects of the Tartessos on water quality can be detected as far away as in sediments in Antarctica, showing that global environmental change is not a 20th century phenomenon.

Therefore, vast clearance of the Mediterranean forests started as soon as in the Late Bronze Age, and continued during the Iron Age until contemporary times. Access to wood in large quantities permitted industries to develop; furnaces for pottery and weapons proliferated. Knossos, the center of the Crete development, was completely deforested by the Late Bronze Age. One city after another overharvested the surrounding landscape, either leading to their decline or to their expansion to less-populated areas. Soil became deprived of vegetation cover, and erosion increased as forested areas were converted to farmland. These were probably the first large-scale land changes in the Mediterranean basin, comparable only to those occurring in the Mesopotamian region at that time. Erosion, flooding and silting of downstream areas were probably common, forcing abandonment of cities and the search for new places to settle.

The pace of environmental change increased during the early years of the current era, when Romans expanded through Gallia, Britannia and Hispania in search for wood, metals and farmland (Perlin 1989). This expansion irreversibly changed the landscape across most of the Mediterranean basin. Romans converted forests into farmland to produce wine, oil, and wheat, and mined metals (iron, silver), stripping large areas to bare soil. They implemented irrigation to cope with the variable Mediterranean rainfall, a trend that was continued later by Muslims of Arab origin. Springs, percolation wells, weirs and reservoirs were built (Figure 14.2), and examples of Roman engineering persist in Mérida (Spain) or Nimes (France).

Figure 14.2: *The aqueduct of Segovia (Spain), a huge waterwork built by the Romans*

The collapse of the Roman Empire brought only a temporary reduction in human population and forest clearance. Modification of the landscape resumed and then increased during Medieval times. Agriculture dramatically changed the landscape; in the Mediterranean basin, agricultural terraces were popular and often favored erosion. Eroded topsoil, transported by the increased frequency of deluges, reached the valley bottoms and filled estuaries.

The spatial scale of transformation became larger and larger. The major aggradation of river channels in Modern times had climatic components but also derived from human activities. As an example, the Ebro River delta increased greatly in size in the Middle Ages and later as a result of forest clearance for livestock. Rivers became a preferred energy source in factories, from smelting to mills (Figure 14.3). As a consequence, the connectivity of streams and rivers was broken by innumerable weirs and dams, resulting in, for instance, reduced salmon runs even

Figure 14.3:
The arms factory in Orbaizeta (Navarre, Spain) is one of the many places where rivers were harnessed to use hydraulic energy. Most streams and rivers were thus dammed and channelized

in rivers where water quality remained high. Although other regions remained agricultural, their forests still decreased and riparian areas were converted for agriculture, leading farmers to protect their lands with dikes and levees. In turn, this disrupted floodplain connectivity, affecting river habitats and their dynamics.

Since the Industrial Revolution, the amount of energy allocated to changing the landscape has increased sharply (Steffen et al. 2007). Such an investment reached its maximum after World War II, and has not ceased since. Its effects increase in extent but vary in intensity. The current percentage of urban, arable and pasture land in the large Mediterranean basins ranges from a mere 15% in the Turkish Gediz to the 69% in the Iberian Guadiana River (Table 14.1). The population density surpasses 200 people per km^2 in the Júcar, Arno and Po (Table 14.1). As a result, most large rivers in the Mediterranean basin are heavily managed, their flow regulated through dams, and natural areas reduced to a minimum. The riparian vegetation in the middle section of the Ebro River presently barely covers 4.5% of the original area, compared to ca. 40% in the 1950s (Ollero 2007).

The 20th century saw a large increase in the human population in the Mediterranean basin, but it was very uneven because the population of rural areas often decreased while that of cities and coastal areas grew steadily. It was also a century of intensified soil use, of increased use of fertilisers and pesticides in agriculture, and of increased pollution in industrial areas. By the end of the 20th century, Mediterranean rivers were among the most degraded in the world. The last few decades have seen improvements in water quality in the richer Mediterranean countries but much less in poorer ones. However, there have been very few advances in other aspects of river conservation, such as the restoration of river habitats.

Although we have focused the history of human river use in the Mediterranean basin, parallel trends in human settlement and subsequent decline in river health have occurred across the world. For example, accelerated soil erosion and sharp declines in river water quality occurred soon after European settlement of many areas of North America, New Zealand and Australia. Only in areas where steep terrain or isolation inhibited population expansion have river landscapes remained relatively intact, and it is these areas that we treasure today as conservation reserves and reference sites for restoration. However, even these are under threat from invasive species and the effects of global climate change.

14.3. Turning the tide: Conserving species and ecosystems

In recent decades, we have become more aware of our impacts on our natural environment. Many of us are motivated to try and protect natural areas and

Table 14.1: *Catchment area, population density and proportions of non-natural land use (urban, arable and pasture) in selected rivers in the Mediterranean basin and Australia*

Iberian Peninsula	**Catchment area (km^2)**	**Population density (hab/km^2)**	**Non-natural land use (urban+arable+pasture)**
Ter	3,010	108	33.7
Ebro	85,362	34	49.6
Júcar	21,208	207	52.3
Segura	19,182	78	55.1
Guadalquivir	57,527	69	63.1
Guadiana	67,048	24	69.1
Tagus	80,600	136	47.6
Mondego	6,670	96	37.4
Duero	97,290	37	56.8
Rhône River basin			
Upper Rhône	8,018	190	19.4
Main Rhône	90,538	141	45.1
Ain	3,713	61	40.3
Saône	29,498	94	63.1
Isère	11,865	82	22.7
Durance	14,322	22	23.5
Balkan region			
Kamchia	5,338	48	48.5
Evros	53,078	69	61.2
Axios	24,604	87	43.5
Evrotas	2,418	30	35.3
Pinios	10,743	54	54.8
Italian Peninsula			
Tagliamento	2,580	50	18.4
Po	73,974	224	49.3
Arno	8,230	243	57.0
Tiber	17,156	238	55.0
Turkey			
Seyhan	20,450	92	32.0
Ceyhan	21,982	91	38.4
Gediz	18,000	113	15.3
Australia			
Swan	80,531	11	86.0
Murray	7,898	2	61.0
Collie	3,771	9	31.0
Blackwood	21,587	1	81.0
Warren	4,395	1	33.0

Source: Data assembled from different chapters in Tockner et al. (2008), and Cooper et al. (2013).

to restore degraded rivers back to some semblance of their original state. This desire to conserve species and ecosystems is not alien to human nature. Many traditional societies set aside reserve areas or protected emblematic animal species for different reasons, including mythological and religious ones. Some of these areas have been crucial to the protection of biodiversity. For instance, the lowland forest of Białowieza, in Poland, has been preserved during centuries as a royal hunting ground, and became home of the last remaining population of European bison, apart from hosting one of the few old-growth forests in the continent, home to unusually high biodiversity. In other cases, the preservation of a species has been related to its economic value. This is the case for several species of the Pacific salmon in regions of the American NW, whose protection also encouraged actions to restore the river habitats. In Australia, indigenous nomadic tribes declared particular edible plants and animals as taboo in different areas, thus protecting food supplies from over-exploitation.

However, often there are not species of apparent economic value (Figure 14.4), traditional approaches to managing wildlife have gone or ecosystems are not favoured by special protection for their unique scenery. Why, then, should we protect those species and ecosystems? *The ultimate reason to move and act is ethical.* We need to move from strictly utilitarian and economic relationships with exploiting natural systems to an approach that recognizes our moral obligations. The American conservationist Aldo Leopold (1949) suggested that extending ethics to environmental issues was both an evolutionary possibility as well as an ecological necessity. Such a land ethic could enlarge the boundaries of human community to include soils, waters, plants and animals. A land ethic could not prevent alteration, management and use of resources, but it would affirm all species' right to continued existence.

Applying this principle of social justice to river ecosystems is to accept that we must find an ethical compromise in our use and conservation of rivers. In practice, this implies that all rivers should have their share of water resources, their dynamics should be maintained, and resident species should not be submitted to chronic stressors. Very often, the fate of water resources transported by rivers is hotly debated among farmers, industry and urban households. In these discussions, the amounts of water (and its quality and timing of availability) that are necessary for river ecosystems to survive and function is neglected. In Spain and elsewhere, it is said, and deeply rooted in the psyche of managers and politicians, that any water reaching the sea is a lost resource. It is also said that maintaining ecological flows is a fantasy in water-scarce regions. Such attitudes stray far from considering ecosystems in the delicate relationships with humans and are unethical, unjust and unsustainable.

Figure 14.4: *Many species, such as the Pyrenean desman* (Galemys pyrenaicus), *are not especially charismatic or economically important. Sadly, these species become the victims of the ignorance and neglect that threatens river biodiversity worldwide. This water mole is the only remaining species of a once diverse genus, and together with the Russian desman* (Desmana moschata), *is one of the only two desman species in the world. Desmans are strictly aquatic insectivores, and the populations of Pyrenean desman are declining swiftly, probably threatened by many of the stressors mentioned in the book, from pollution to disruption of connectivity. Because this species has little appeal to the general public, its prospects for survival are dim*

These attitudes are also economically flawed. They overlook the fact that river ecosystems provide essential services for humans, and that these services provide significant economic gains. Destroying or impairing these ecosystem services is detrimental to our own interests as well as those of many other companion species. However, we need to be aware that most current dangers to river biodiversity and ecosystem functioning are subtle, may remain unnoticed or may be hidden by pervasive but ill-informed economic reasoning. Several chapters in this book have shown that most disturbing alterations in river systems come from sediment disruption, riparian modification and water abstraction. Other chapters show that dissolved elements such as nutrients and pollutants produce chronic effects on the biota and alter their performance in the ecosystem. This opens the gate to the entry and settlement of invasive species, as well as favoring the disappearance of less-tolerant (usually native) species and impairing their genetic richness and connectivity.

Conservation has direct – even if not immediate – benefits for humankind, and our future generations will appreciate this. The conservation ethic is an attitude that goes beyond the immediate benefit to be obtained. The ecosystem goods, including species and the ecosystems that host them, are our common heritage. Accepting this heritage requires an ethical position because without it there are not sufficient scientific reasons to justify species' and ecosystems' conservation.

Conserving rivers requires a strong commitment from all sectors of our society. Our contribution to preserving them begins by recognizing that we both use and enjoy rivers and their benefits

The closing chapters of this book show that agreements can be reached between scientists and managers, and that successful river conservation may come together through Strategic Adaptive Management where all users become actively engaged towards a common goal.

Overall, our book has a number of key messages in river conservation (Box 14.1). These messages revolve around the theme that river conservation is not only ethically correct but also economically sensible in the longer term. Action is urgently needed, along with a change in public attitude and behavior. Prevention (i.e. conservation) is always better, cheaper and more sustainable than cure. Although we now face global problems of climate change and population growth against a backdrop of economic uncertainty and unequal access to the world's resources, we must strive to use our technology and intelligence to improve the quality of life of all humans and our supporting ecosystems, including rivers. We are optimistic that this is possible but we need everyone to share our vision, optimism and knowledge to bring this about.

14.4. How *you* can make a difference

We are optimistic about the future. We believe there is an accelerating change in our collective social tradition to move away from individualism, consumerism and nationalism, and embrace a broader global view of our situation as a species within a complex ecosystem. Part of this stems from wider public and political recognition of global environmental issues such as climate change and their economic and social implications. Part also stems from the relatively rapid change in public attitudes towards the environment. For example, in the 1970s, environmentalists were considered to be unconventional oddities and the name "greenie" was used in a derogatory sense. Today, major companies clamour to prove their "green" status in an economy where consumers seek products that can be shown to be derived in ways that are environmentally sustainable.

Often, many of us have a feeling of powerlessness and despair when we learn more about the state of the world's rivers and their ecosystems. But this should not paralyse us or prevent us from trying to do something about it. As individuals and groups, we can help to conserve rivers and their dependent ecosystems. Actions range from relatively simple everyday changes in behaviour through to major involvement in river conservation campaigns and restoration activities (Box 14.2). It is simply a case of making a personal commitment to act ethically.

Box 14.1

Key messages in river conservation

River conservation is not only about preserving scenic landscapes or beautiful fishes. Degradation of river condition exerts a harsh toll on human society. For example, poor river water quality is a major source of problems, from public health to economic issues.

In addition to these utilitarian values, we hold that river ecosystems have intrinsic values and, as such, humans should not destroy these values simply by neglect. As a society, we have a duty to pass on to our heirs the environmental wealth, including natural rivers, that we inherited from our ancestors.

There is no time to lose. Our generation, and at most the next, will be the last ones with the capacity to conserve a large fraction of current biodiversity. Channel form and water chemistry might be restored in the future, but once we lose species, they are gone forever.

It is always easier to prevent something from breaking down than to fix it after it is broken. For rivers, it is far easier to conserve than to restore them. This means that we should increase our efforts to conserve the few remaining near-natural rivers instead of trying to restore them once they have been degraded.

Even under the most environmentally friendly scenario, Earth will undergo profound changes by the end of the 21st century. Thus, we must devise ways to conserve and restore rivers, although their drainage basins will be far from "natural". Therefore, we must seek ways to conserve or restore functional ecosystems in landscapes that are no longer pristine. Recovering the "river territory" (the land adjacent to the channel where the river is free to adjust its dimensions and to migrate) will be one key step.

Our society will have to find the right technology to face current and future threats. Any vision of "returning to our ancient relationship with Mother Nature" cannot work, simply because we cannot sustain the current human population with ancient production methods. Technology is necessary, although it is a curse and a blessing at the same time. No technology is totally green, as shown by the example of hydropower, but a world without technology is not the solution to our problems.

In many fields, there is urgent need for more information. One example is the proliferation of new pollutants. We should apply the precautionary principle and not adopt a technology until we know how to manage the risks that it will create. This is not a new policy. For instance, in the issue of urban waste, in the last few decades we have changed our policies from total neglect to very strict controls on producing and dumping this waste. Other stressors, such as pharmaceutical drugs, will have to follow the same path.

Box 14.2

Ten ways that *you* can help conserve our rivers

1. **Be frugal with water**
   In your everyday use of water, even when it appears plentiful, try and reduce your use. Stand under the shower for less time and turn it off while you soap up. Fix dripping taps immediately. Install low-flush toilets and low-flow showerheads. For more ideas, see http://water.epa.gov/polwaste/nps/chap3.cfm

2. **Protect water quality at home and at work**
   We all live in catchments. Excessive use of pesticides and herbicides in our gardens can leach pollutants into waterways. Poorly maintained vehicles leak oil and noxious materials. Dispose of toxic materials (e.g. house paint) appropriately. Reduce paper use and recycle everything because production of material involves water. Never dump rubbish in waterways, even when they are dry.

3. **Help clean up local waterways and riparian zones**
   As individuals, we can pick up rubbish and be alert to sources of pollution. As groups, we can arrange "river clean up" days which also help build collective community support for river conservation and protection. If you see illegal dumping or unmonitored sources of pollution from industry, let your local council know.

4. **Volunteer to work on river conservation projects**
   Groups achieve more than individuals. Everyone brings different skills and strengths to a group, and you may also be a natural leader. Projects such as tree planting, bank stabilisation and removal of noxious weeds from the riparian zone help conserve rivers. Check council regulations before any activity and seek professional advice from natural resource agencies and local environmental groups.

5. **Be informed about environmental issues**
   Many local councils have excellent resource material about protecting and conserving local rivers. If your council does not have information on your river, encourage them to obtain it. Your requests will stimulate greater environmental awareness and responsibility in local government. Many websites (see later) also cover this topic.

6. **Join river conservation and protection campaigns**
   Campaigns involving media coverage and collective activity (e.g. river restoration works) help raise public awareness of river conservation. Residents living near rivers should be especially aware of their ethical responsibilities. Try and involve local schools in hands-on programs and help teachers promote river conservation in school curricula.

7. **Donate money or time to dedicated conservation groups**
   Most conservation groups rely on volunteers. They also need money to support campaigns, maintain offices and provide materials for river restoration activities. Every little bit helps and it is an excellent way to learn more about environmental issues.

8. **Share your knowledge and ethical beliefs in river conservation**
   Many of the major improvements in human rights became consolidated by public awareness followed by social endorsement. Being willing to share your views on the virtues of conservation helps empower others to feel the same way. Be optimistic.

9. **Press your elected person to act in favor on river conservation**
   Too often citizens have low participation in local politics. It is our right and our duty to talk to elected persons (from council representatives to national ones), and to ask them to work in favor of the environment. If politicians feel pressure, they are more likely to have environmental issues in mind in their everyday work.

10. **Think about nature conservation when going to vote**
    In most countries, environmental issues have little weight in the political agenda, and people tend to vote for one party or other following more general criteria. We think environmental issues should be more central to the political debates, and be incorporated in the main ideary of political groups. This will not occur unless conservation becomes a key element when deciding who to elect.

The most important point is to become actively involved. We hope that you have become motivated and inspired by the chapters in this book. Although there are many problems threatening our rivers, there are also solutions. All of these need you to become involved in some or all of the activities outlined in Box 14.2. They also need a change in cultural attitudes so that everyone accepts the virtue of conservation. We must face up to our ethical obligations. It is socially just that we share our water equitably and sustainably with the environment and other users. We must pass on to our children rivers that are as healthy or even more so than the ones we inherited. Will you accept this conservation challenge?

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14.5.1. Useful websites

The Nature Conservancy protects nature and preserves life across over 30 countries and their website has details about regional projects and specific activities in conservation, including river and riparian zone conservation. [http://www.nature.org/ourinitiatives/index.htm]

The River Network provides helpful hints on ways to conserve water and use it effectively. It also has useful links to current campaigns on river conservation (mainly in the US) and an active blog on many topics ranging from river protection laws to restoration strategies. [http://www.rivernetwork.org/resource-library/how-conserve-water-and-use-it-effectively]

Conservation Commons is a website that collates open access to data, websites, information and knowledge on general biodiversity conservation, including that of rivers and streams. [http://www.conservationcommons.net/]

Water Culture has a readable review of the general ethics of water use, linking cultural aspects with decisions about how we use water in domestic, agricultural and industrial situations. [http://www.waterculture.org/Ethics_of_Water_Use.html]

Water Footprint Network presents the context and a "calculator" for measuring our "water footprint" – the direct and indirect usage of water for the goods and services we use in everyday life. [http://www.waterfootprint.org/?page=files/home]

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