15 How We Manage Rivers, and Why

The world's ecosystems have been extensively altered throughout the age of the Anthropocene, river ecosystems perhaps most of all. With nearly half of global river volume moderately to severely impacted by dams and other waterworks (Grill et al. 2015; Lange et al. 2018), over half of available freshwater runoff captured for human use (Jackson et al. 2001), one-fourth of the global sediment load trapped behind dams before it reaches the oceans (Vörösmarty and Sahagian 2000), and several of the world's great rivers no longer flowing to the sea during dry periods (Postel 2000), the physical evidence is overwhelming (Best 2019). Although these are amongst the most dramatic examples of anthropogenic impact, other pressures resulting from human actions are of major importance. Broad categories of threats described earlier include pollution, flow modification, habitat degradation, over-exploitation, species invasions, and climate change (Table 1.1).

Sadly, there is no shortage of evidence that biological assemblages and ecosystem processes have been profoundly altered by these pressures acting alone or, often, in combination. Freshwater ecosystems are among the most threatened and their communities among the most imperiled on Earth. By area, freshwater ecosystems occupy less than 1% of the Earth's surface yet contain 10% of all known species, including about one third of all vertebrates (Strayer and Dudgeon 2010). A compilation of geographical range data for 7,083 freshwater species of mammals, amphibians, reptiles, fishes, crabs, and crayfish found that almost one in three is threatened with extinction world-wide (Fig. 15.1). In addition, all six groups exhibited a higher risk of extinction than their terrestrial counterparts, and extinction risk was estimated to be higher in lotic habitats than wetlands and lakes (Collen et al. 2014). Urgent and concerted global action is needed to stem the loss of freshwater biodiversity while there is still time (Tickner et al. 2020).

The primary purpose of this chapter is to describe what can be done to reverse this harm. Specific actions will depend on many variables associated with local circumstances. Is a particular location and stream or river best viewed within the framework of repair, restore, or protect (Fig. 1.10)? What direct and indirect economic values are at issue, and what aesthetic and natural values? What institutions, and what policy and legal frameworks will influence the process, and what is the level of community engagement with the resource? While many factors will influence how best to manage a riverine ecosystem to improve its condition, we believe the starting point should be at a higher level. Why should we endeavor to repair, restore, and protect rivers, and what should be our over-arching goal?

Most of us have an intuitive idea that streams and rivers benefit humans. They are a source of drinking water and harvestable fish, of hydropower and irrigation water when harnessed by dams and canals, useful for navigation, and have functioned as a defensive barrier for ancient cities. Their floodplains absorb flood waters, slowing downstream passage while capturing sediments and nutrients that enrich the floodplain's agricultural potential. In addition to these tangible benefits, running waters have aesthetic values that include the pleasures people experience from fishing, paddling, or strolling along a riverbank, but can extend much further into the spiritual realm. Beyond the sciences of hydrology, geomorphology, ecology, and other disciplines that contribute to our understanding of rivers, running waters have served as muse and metaphor for philosophers, poets, and humanist writings about people and nature. Before concluding this book with an exploration of how scientists, citizens, managers, and decision-makers can most effectively work to improve the status of rivers, we begin with the most important question: why should we do so?

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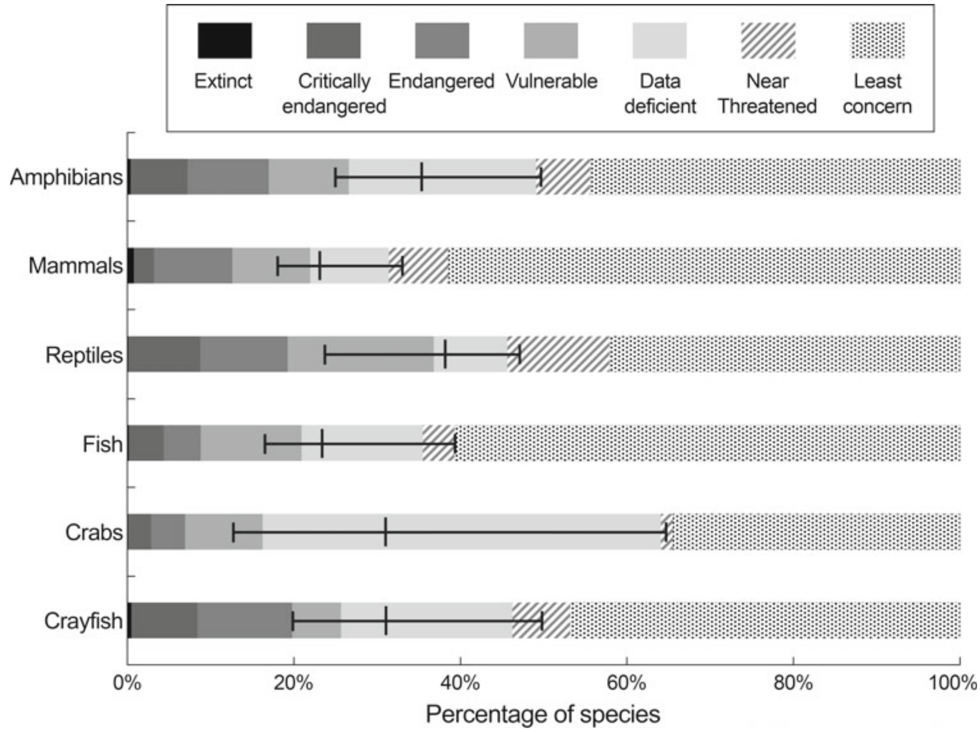

Fig. 15.1 Extinction risk of global freshwater fauna by taxonomic group. Central vertical lines represent the best estimate of the proportion of species threatened with extinction, with whiskers showing confidence limits. Data for fish and reptiles are samples from the respective group; all other data are comprehensive assessments of all species (n = 568 crayfish, 1191 crabs, 630 fish, 57 reptiles, 490 mammals and 4147 amphibians). Solid colors are threatened species, from left to right: black, extinct; darkest grey, critically endangered; mid-grey, endangered; light grey, vulnerable; lightest grey, data deficient. Patterned bars are non-threatened species: hatched, near threatened; dotted, least concern. (Reproduced from Collen et al. 2014)

15.1 Benefits from Intact Rivers

15.1.1 Ecosystem Services

Recent years have seen the development of conceptual frameworks for assessing the value of an ecosystem, as well as increased application of economic tools in an effort to monetize different benefits that an ecosystem provides. The supply of ecosystem services, defined variously as the goods and services that an ecosystem provides free of charge, and as the benefits that people receive from ecosystems (Millennium Ecosystem Assessment 2005), is widely recognized as a potent framework to justify management and restoration actions. Ecosystem services (ES) often are grouped into four broad categories. Provisioning services include the production of directly consumed resources, such as fish, drinking water, and hydropower. Regulating services are the benefits obtained from regulating processes, including waste decomposition and water purification, flood control, and pest suppression. Supporting services (sometimes combined with habitat services) include basal resources, nutrient and other biogeochemical cycles, degradation of organic wastes, and species' habitat. Cultural services include educational, recreational, aesthetic, and spiritual benefits.

Recent work favors what is known as the ES cascade, a useful framework for operationalizing ES quantification by breaking the concept into measurable entities (Boerema et al. 2017). The cascade framework links natural systems to aspects of human well-being, following a pattern similar to a production chain: from ecological structures and processes generated by ecosystems, to the services and benefits eventually derived by humans (Haines-Young and Potschin 2010). For example, the existence of floodplains and riparian wetlands in a catchment may dissipate the energy and slow the passage of a flood. This function of the ecosystem connotes its capacity to do something that is potentially useful to people, and so can be considered an ecosystem service. The human benefit of this ecosystem service will depend on the extent of harm that may occur due to flooding, and its value can be estimated by methods described below, such as peoples' willingness to pay to maintain this service. A central point is that ecosystem services exist (are realized) when some benefit accrues to people. That benefit can be experienced directly at the location where the service is realized, but benefits also can be experienced at a distance, for example when clean drinking water is the result of land management in headwaters, or when someone in North America gains satisfaction from the existence of river dolphins in the Amazon River.

Two important ideas flow from explicit consideration of ecosystem services. First, human actions that degrade ecosystem condition and function may compromise the ecosystem's ability to deliver ES. An ES framework makes explicit that benefits may be lost due to environmental stressors. It also provides a way to message and quantify the benefits gained under ecological restoration. Second, the recognition that ecosystems provide not one but many benefits raises the question of the inter-relationships among multiple services. Ecosystem properties, processes, and components that are the basis of service provision will be affected, usually adversely, by myriad human activities and, hopefully more positively, by management and restoration aimed at reversing environmental degradation. As a consequence, individual services as well as the complete bundle of services provided by that ecosystem will change. In some instances, management intended to improve some ecosystem service may benefit other services as well, in what is known as co-benefits or a "win-win" outcome. Protecting a wetland provides both wildlife habitat and improved water quality for human uses. In contrast, management intended to improve one ecosystem service may adversely affect another, indicating tradeoffs in choices and outcomes. Providing for a spring flood pulse may conflict with ensuring that enough water remains in the river for year-round navigation. Similarly, an intense storm event may replenish drinking water supplies for a given jurisdiction, but flood neighborhoods downstream. Conflicting outcomes can be common, especially in urbanizing watersheds, and the distribution of services and disservices experienced by a community are often influenced by income and demographics (Keeler et al. 2019).

The total value of a river is the sum of its direct and indirect uses, and its non-use values, which require different methods for their estimation. Direct-use value refers to the value of some ecosystem product as a commodity that can be sold in a market at a known exchange price. Harvested fish and electricity generated from hydropower are examples. Indirect use value stems from benefits to human society from indirect utilization of ecosystem services. Flood protection afforded by floodplains, natural water filtration, and carbon sequestration are examples, as are recreational uses and activities. Additional values that have been recognized include preserving the option to utilize ecosystem services in the future (option value), satisfaction that an ecosystem exists (existence value), recognition of the welfare the ecosystem may give other people (altruistic value), and preserving the ecosystem for future generations (bequest value).

When an economic value cannot be derived from existing markets, methods to monetize indirect and non-use services include stated preference, revealed preference, and benefit transfer. Stated preference methods such as contingent valuation ask people for their willingness to pay for a certain ecosystem or service, typically with a survey presenting choices or alternative scenarios. Revealed preference methods relate peoples' willingness-to-pay for a service to their actual expenditures or the value of some market good or service. Hedonic pricing estimates a value for some ecosystem service such as water quality from its statistical relationship with the price of a good for which a market actually exists, such as waterfront housing. A number of studies have shown that water views and proximity to shoreline is highly desirable in residential housing markets. Sales data on land parcels adjacent to the Neuse River in North Carolina, US, established that a riparian property generally commands a significant premium compared to an otherwise equivalent property (Bin et al. 2009). Travel cost methods use gas mileage costs, entry fees, on-site expenditures, and outlays on recreational equipment as substitutes for the market price of some environmental good or service. Recreational fishing in the rivers of the Pantanal region of South America draws several tens of thousands of Brazilian anglers for typically week-long trips (Shrestha et al. 2002). Travel costs determined by angler survey of \$86–\$140 per day are high relative to similar estimate for the US of around \$33 per person day in 1996 dollars. Aggregate value of recreational fishing in the Brazilian Pantanal ranges from \$35 to \$56 million.

Stated and revealed preference methods are both widely used. Revealed preference has the advantage that it is based on estimates of actual dollars spent. Stated preference methods have the disadvantage that it is unclear whether peoples' expressed willingness-to-pay translates into actual dollars, but have the advantage that they can be applied to non-use values such as existence values of fish and wildlife (Bergstrom and Loomis 2017).

The transfer of estimated benefits from one site to another, known as benefit transfers, is widely used to argue for site protection in environmental decision-making (Plummer 2009; Richardson et al. 2015). In benefit transfers, a single value from an empirical study of a site, or the mean from multiple study sites, is used to provide a value estimate for that ecosystem service at similar sites over a large region. Often used to reduce effort and expense in analyzing options, its greatest potential pitfall is the assumption of correspondence among locations (Plummer 2009). A preferred alternative is a benefit function that relates an estimated willingness-to-pay to a set of site characteristics, including its socio-economic setting. When a benefit function is based on multiple sites exhibiting a range of conditions, a more robust benefit transfer can be accomplished by measuring the function's variables at a new site and evaluating the function at those values.

Stated willingness-to-pay methods have shown that people value ecosystem protection for some non-use amenity, even from a distance. Using contingent valuation to assess respondents' willingness-to-pay for removing two dams to restore the ecosystem and its anadromous fishery of the Elwha River in Washington State, US, Loomis (1996) estimated aggregate benefits to residents of the state at $138 million annually for 10 years. Reasoning that restoration of a river in a national park and increases in salmonid populations are public goods available to all, the survey was also sent to residents throughout the US. Results indicated that the general public would be willing to pay between $3 and $6 billion, revealing the substantial nonmarket value of removing old dams to restore salmon and steelhead runs in the Pacific Northwest. Asked their willingness to pay for five ecosystems services that depended on a trade-off between instream flows versus off-stream uses, residents along the Platte River, Colorado, US, found that, on average, individuals would pay an additional $250 annually via a higher water bill (Loomis et al. 2000). When extrapolated to the population living along the river, ES values exceeded costs of alternative conservation efforts. Using geo-tagged photographs as a proxy for recreational visits to lakes in Minnesota and Iowa, US, Keeler et al. (2015) showed that number of visits increased with improved water clarity. Recreational lake users were willing to travel farther and incur increased travel costs to visit lakes of greater clarity, a finding consistent with stated preference studies.

Despite the potential for estimates of non-market value to benefit environmental decision-making, actual monetization studies still are relatively few. A literature search for studies of river restoration that quantified and valued one or more ecosystem goods or services found 32 examples, including 24 in the United States, six in Europe, one in Mexico, and one in China (Bergstrom and Loomis 2017). Restoration of fisheries accounted for two-thirds of the examples, and included studies focused on threatened species, native species, and recreational fishing. Estimation methods included stated preference by contingent valuation and choice experiments, revealed preference including hedonic price and travel cost estimates, and benefit transfer. Stated preference estimates were most common. Willingness-to-pay estimates for river restoration increased with length of river restored and number of goods and services valued. The authors inferred that these valuation estimates were used primarily as background information, although in some cases they entered more directly into decisions.

While advances in the valuation of ecosystem services have provided new tools for capturing the worth of ecosystems and communicating this to an audience more familiar with monetary valuation, such approaches have yet to capture all dimensions of value. Chan et al. (2012) propose a typology that recognizes eight dimensions of value, including self- vs other-oriented, physical vs metaphysical, and anthropocentric vs eco-centric, among others. Reliance on an ES perspective raises questions about the judgments we make in assigning value to nature, and poorly resolved ethical concerns concerning the relationships between humans and non-human nature (Jax et al. 2013). By emphasizing monetary valuation, the result is that other, less tangible non-use values are marginalized in decision-making. Some experiences of nature are especially intangible, such as a love of nature or explicit, spiritual connection with some natural feature. Few would advocate monetizing a sacred site in order to negotiate a trade-off with resource extraction. In a similar vein, all ecosystems, including rivers and streams, have intrinsic value that many people will feel uncomfortable expressing in currency. This segues to a second, certainly co-equal answer to why we protect rivers, which we will call rheophilia.

15.1.2 Rheophilia

Are rivers an amenity, that we may utilize as needed, or even replace with manufactured alternatives such as de-salinized drinking water, fish production by aquaculture, and designed streams flowing through designed landscapes? Or do streams and rivers play a deeper role in supporting human well-being, and if so is that role enhanced by, or does it even require, the opportunity to experience diverse river settings in as near-natural a state as we can achieve? And if the latter is closer to what we humans desire, how can we best make the case for the required effort to protect rivers? One line of argument, born of the need to demonstrate human benefits in terms of economic value, is described in the above discussion of ecosystem services. A second line of argument, only partially captured by cultural ES, is expressed beautifully by the title of a conference address by Luna Leopold in 1977, *A Reverence for Rivers* (Leopold 1977), and of a book by Kurt Fausch, *For the Love of Rivers* (Fausch 2015). These are in the tradition of scholarly explorations of the human basis for the love of nature and its restorative benefits, developed in the writings of E.O. Wilson (*Biophilia*, (Wilson 1984) and Rachel and Stephen Kaplan (*The Experience of Nature: A Psychological Perspective*, (Kaplan and Kaplan 1989). Nor should this perspective require academic scholarship, one might argue, for it also is captured in the frequent mention of running waters in literature, art, and song. Perhaps restoring rivers is not only to improve the condition of the river ecosystem, but is equally or even more about enhancing the river's restorative capacity for human well-being. Cultural ecosystem services recognize this, but to date have had limited success in capturing these values, which to us seem best viewed as a separate and equal domain.

Perception of the attractiveness of river and riparian scenes can be determined from preferences expressed for photographs that depict different settings. Using a questionnaire to assess reactions to 20 pictures of rivers from over 2,000 students across ten countries, in terms of naturalness, danger, aesthetics, and need for improvement, Le Lay et al. (2008) found that a preference for mountain streams with turbulent flow and boulders was common to all. Whitewater and scattered large boulders characterized the most aesthetic and natural riverscapes, whereas rivers characterized by extensive gravel bars, narrow bands of water, and large amounts of wood were considered the least attractive. Surveys of the public's perception of river corridors in southern England and Wales found strong preferences for mature, sinuous rivers with natural channels and vegetated banks (House and Sangster 1991).

Interestingly, despite much scientific evidence of the benefits of wood in streams, several studies have found negative public perception of wood in rivers. River channels with woody debris can be considered less aesthetically pleasing, more dangerous, and needing more improvement than those without wood (Chin et al. 2008). Cross-cultural comparisons indicate that perceptions of riverscapes are influenced by cultural setting. Respondents from some European countries and the US had a negative perception of regulated rivers, and a positive perception of wood within streams; participants from India, China, and Russia were the opposite (Le Lay et al. 2008). Such differences may stem from many elements of lived experiences, culture, and local environmental history, as well as from education and its communication.

Perception of the attractiveness of river sounds can be assessed using audio-recordings that compare urban to natural or park-like settings. A survey of preferred sounds in two squares in the city center of Sheffield, England, found a preference for natural sounds and especially for the sound of water in park fountains, making soundscape an important element of the design of urban spaces (Yang and Kang 2005). Subjects exposed to stress (given three seconds to determine if an equation was correct or false) and monitored for physiological response exhibited faster mood recovery when experiencing nature sounds (a fountain and bird calls) in comparison with urban noise such as traffic (Alvarsson et al. 2010). When presented with sound and image combinations representing a stream, a village, a quiet park, a busy park, and a residential neighborhood, subjects expressed an overall preference for natural and rural over urban and man-made scenes. The most highly rated combination was the sound and image of a stream (Carles et al. 1992).

Beyond an expression of preference, studies show that visual images of natural environments facilitate attention restoration, improve mood, and can more generally enhance health. There is evidence that interacting with nature has cognitive benefits, in part because of the attention-capturing distractions of navigating an urban environment in comparison with a more natural setting. Participants assigned to walk for 50 min in a large urban park performed better on a cognitive task than others who walked in the downtown area of a city (Berman et al. 2008). A second experiment found that participants performed better at more complex attentional functions when viewing photographs that depicted scenes from nature in comparison with city scenes.

We cannot do justice here to a humanist perspective on flowing waters, revealed in art, poetry, and great works of literature, but we would be remiss not to mention it. The Hudson River School was a mid-19th century group of American landscape painters whose work drew inspiration from the Hudson River and the surrounding area. Known for their realistic, detailed, and sometimes idealized portrayal of nature, their paintings often juxtaposed peaceful agriculture and the remaining wilderness, or portrayed an idyllic scene of still-pure nature. Celebrated landscape artists Frederic Edwin Church and Albert Bierstadt were a second generation of this school, and running waters were central to some of their most famous paintings, including Church's *Niagara*, *Morning in the Tropics*, and *Heart of the Andes*. Looking to a different culture and a different time, *Along the River During the Qingming Festival* (the *Qingming Shanghe Tu*) painted by the Song dynasty artist Zhang Zeduan (1085–1145) depicts the daily life of people and the landscape during a period of the Song Dynasty. Said to celebrate the festive spirit and worldly commotion at the Qingming Festival, this is considered to be the most renowned work among all Chinese paintings.

Many fine works of literature, both fiction and non-fiction, draw inspiration from rivers. Mark Twain's great novels, *The Adventures of Tom Sawyer* (1876) and *The Adventures of Huckleberry Finn* (1884) surely drew their river settings from Twain's years as a river boat pilot, which also formed the basis for his *Life on the Mississippi* (1884). Henry David Thoreau, an American philosopher of nature best known for *Walden, Or Life in the Woods* (1854), earlier published *A Week on the Concord and Merrimack Rivers (*1849), describing his 1839 hiking and boating trip with his brother through parts of Massachusetts and New Hampshire. And in one of the finest short stories about fishing every written, T*he Big Two-Hearted River* (1925), Earnest Hemingway describes not just the dedicated chase after a large trout, but the healing and restorative power of nature following the devastation of the First World War. For more contemporary American writings, an admittedly selective short list would include *A River Runs Through It*, Norman Maclean semi-autobiographical account of coming of age in an early 20th-century Montana family in which "there was no clear line between religion and fly fishing"; *River Horse*, William Least Heat-Moon's account of travelling across America not by road but by water; and *Desert Solitaire*, Edward Abbey's vignettes of river running and explorations in the American southwest. There is no shortage of mention of rivers in poetry and song, and here we highlight just one example. Any who has travelled a river by canoe cannot help but be moved by "The Song My Paddle Sings" by the Canadian poet Pauline Johnson (1862–1913), also known as Tekahionwake, the daughter of a Mohawk Chief and a woman of English parentage.

In the end, the answer to the question, "Why protect rivers?" is straightforward. River ecosystems provide many benefits to humans, and both our understanding and our ability to quantify these benefits are advancing rapidly. Arguably, however, this is the lesser of two rationales. As the Senegalese forest engineer, Baba Dioum, said in a 1968 presentation to the International Union for the Conservation of Nature, "In the end we will conserve only what we love, we will love only what we understand". The ocean explorer Jacques Cousteau summed it up more succinctly: "people will only protect what they love". Shōzō Tanaka, considered to be Japan's first conservationist, said: "The care of rivers is not a question of rivers but of the human heart". Protecting rivers ultimately is the responsibility to protect what we love.

Understanding the why gives motivation and urgency. The how of repairing, restoring, and protecting rivers blends river science, human perceptions and beliefs, socio-economics, politics, and much more. The following sections attempts to inform readers of some of the major approaches and challenges.

15.2 Goals in River Management

Setting realistic goals for river management must, as a beginning point, be guided by some appraisal of threats and opportunities. There must also be some level of societal support, or a plan to garner support. Individual streams and rivers span an enormous range of settings and challenges, from highly compromised systems, to those where restorative actions hold great promise, to still others where protection against future threats may suffice to preserve them in a near-natural state. Specific objectives likely will depend very much on both the condition of the river system and how societies view its uses and values.

Over time, perspectives on river management have shifted from a more limited focus on meeting human needs while attempting to mitigate environmental costs, to one that emphasizes sustained human benefits, including water for direct human use and water to support other services supplied by healthy ecosystems. The terminology of river management can be distracting, as management actions can be described as restoration, rehabilitation, and improvement; and integrated river basin management (IRBM) is interchangeable with integrated watershed and catchment management. More generally, these approaches fall under the rubric of ecosystem-based management and adaptive management, ideas whose ascendency dates to the 1990s. Ecosystem-based management advocates a holistic approach that recognizes the full array of interactions within an ecosystem, including people and their activities, and the need for cooperative management over large jurisdictional areas (Slocombe 1993). Adaptive management is an integrated, interdisciplinary approach that emphasizes on-going cycles of learning through management interventions, whether they succeed or fail, and the harmonizing of environmental and societal goals as the guiding framework (Walters and Holling 1990).

Management actions aimed at improving rivers increasingly emphasize a holistic approach that attempts to create or maintain some aspect of river form and function that aligns with hydrologic, geomorphic, and ecological processes (Wohl et al. 2015). This can be accomplished by relieving pressures that degrade and harm a river system, thereby promoting natural recovery; and by active measures to assist recovery that may include dam removal, addition of habitat elements, control of an invasive species, ensuring environmentally beneficial flows, and many more such actions. More generally, it includes diverse management activities intended to improve the hydrologic, geomorphic, and ecological processes within a degraded watershed and replace lost, damaged, or compromised elements of the natural system. Other rivers of the same region and approximately the same environmental setting, that are relatively undisturbed, often serve as the benchmark, and the aspiration of achieving healthy ecosystems is at the forefront. Unfortunately, however, coordination of multiple projects throughout a catchment, and consideration of pressures arising at large spatial scales, too often are ignored (Bernhardt et al. 2007; Feld et al. 2011; Friberg et al. 2016).

From the 1980s onward, ecological restoration came into wide use to describe management activities intended to restore damaged ecosystems to a more natural, undisturbed state. Characterized by a more explicit pairing of science and practice, and by goals focused more strongly on recovering historic form and function, this perspective rapidly took hold in river management, resulting in a dramatic rise of projects characterized as river restoration (Bernhardt et al. 2005). This has resulted in a rapidly expanding literature that describes individual projects as restoration work. In addition, there has been much discussion concerning the feasibility of attempting a return to pre-disturbance condition, as well as much analysis of their success or lack thereof, both discussed below. In this chapter we prefer to lump all such activities under the label of management, but where researchers describe their work as restoration, we do as well.

River restoration projects have a wide range of objectives. Based on over 37,000 projects compiled from governmental databases, gray literature, and contacts from seven regions of the coterminous US, Bernhardt et al. (2005) identified 13 categories of river restoration, together with median cost and typical activities or measures taken (Table 15.1). From a database of 813 hydromorphological river restoration projects mostly from Europe, Friberg et al. (2016) identified 53 specific measures grouped within 8 categories: water quantity, sediment quantity, flow dynamics, longitudinal connectivity, in-channel habitat, riparian zone, river planform, and floodplain. Habitat improvements were common, including the removal of artificial embankments, the addition of large wood, and the provision of spawning gravel. A compilation of information on 644 restoration projects from 149 published studies that provided quantitative information on the effectiveness of restoration projects found the most common objectives to be related to increasing biodiversity, stabilizing channels, improving riparian and in-stream habitat, and improving water quality (Palmer et al. 2014, Fig. 15.2a). Methods used were dominated by physical manipulations, such as moving channels laterally, adding sinuosity, or raising/lowering the bed or floodplain for reconnection; and addition of in-stream structures, such as boulders, logs, and gravel (Fig. 15.2b). From these and other reviews of restoration activities it is

Table 15.1 Common river restoration goals and activities, following Bernhardt et al. (2005) and Wohl et al. (2015). Although these categories are not fully independent, they are common rationales for most restoration projects. Projects aimed at improving water quality, riparian management, and habitat improvements were amongst the least expensive and most frequently carried out in the analysis of Bernhardt et al. (2005). Stormwater management, floodplain restoration, and dam removal were more expensive and less common

Goal	Description of activities
Esthetics/recreation/education	Activities that increase community value: use, appearance, access, safety, and knowledge
Bank stabilization	Practices designed to reduce or eliminate erosion or slumping of bank material into the river channel; this category does not include stormwater management
Channel reconfiguration	Alteration of channel geometry, planform, and/or longitudinal profile and/or daylighting (converting pipes or culverts to open channels); includes meander restoration and in-channel structures that alter the thalweg
Dam removal/retrofit	Removal of dams and weirs or modifications/retrofits to existing dams to reduce negative impacts; excludes dam modifications that are simply for improving fish passage
Fish passage	Removal of barriers to upstream/downstream migration of fishes; includes the physical removal of barriers, construction of alternative pathways, and migration barriers placed at strategic locations along streams to prevent undesirable species from accessing upstream areas
Floodplain reconnection	Practices that increase the inundation frequency, magnitude, or duration of floodplain areas and/or promote fluxes of organisms and materials between channels and floodplain areas
Flow modification	Practices that alter the timing and delivery of water quantity (does not include stormwater management); typically but not necessarily associated with releases from impoundments and constructed flow regulators
Instream habitat improvement	Altering structural complexity to increase habitat availability and diversity for target organisms and provision of breeding habitat and refugia from disturbance and predation
Instream species management	Practices that directly alter aquatic native species distribution and abundance through the addition (stocking) or translocation of animal and plant species and/or removal of exotic species; excludes physical manipulations of habitat/breeding territory
Land acquisition	Practices that obtain lease/title/easements for streamside land for the explicit purpose of preservation or removal of impacting agents and/or to facilitate future restoration projects
Riparian management	Practices that improve riparian and bank condition including riparian buffer creation and maintenance, revegetation, eradication of weeds and nonnative plants, livestock exclusion
Stormwater management	Practices intended to reduce stormwater runoff at source and reduce hydrologic scouring by means of rain gardens, pervious pavers, holding/retention ponds; constructed wetlands lower in watershed to filter sediments and nutrients
Water quality management	Similar to above but including water treatment infrastructure and regulatory control of pollutants, as well as landscape-scale best management practices to reduce runoff and capture sediments and nutrients

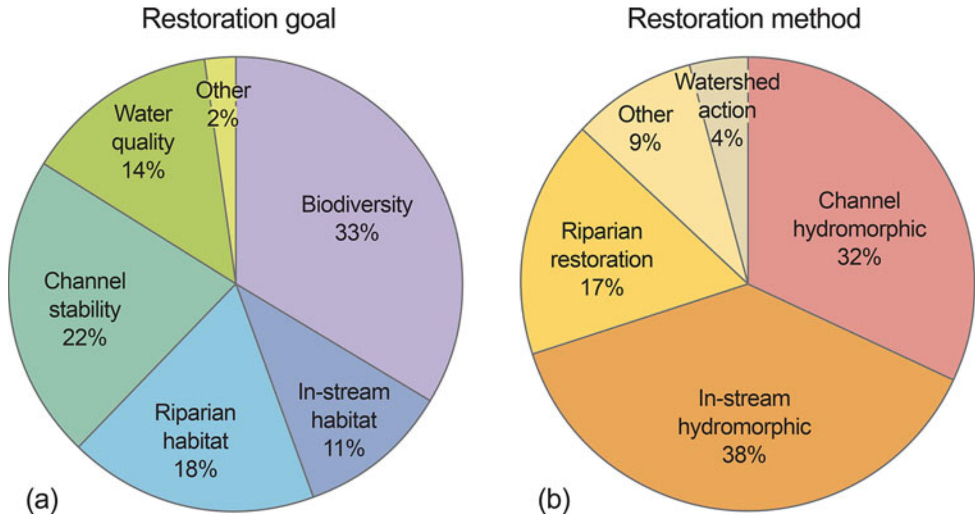

Fig. 15.2 Summary of the most common restoration goals and implementation methods for 644 river or stream restoration projects; values are percentages of projects using a goal or method. (**a**) The primary goal of each restoration project, (**b**) restoration methods depending on how the project was implemented. Channel hydromorphic projects involved reconfiguring the channel, such as moving it laterally, adding sinuosity, or raising/lowering the bed or floodplain for reconnection, and they often included addition of in-stream structures, such as boulders, logs, and gravel; in-stream hydromorphic projects were less intensive projects that involved only manipulating in-stream structures, adding large woody debris, armoring the bank, or creating artificial riffles without major channel excavation or reconfiguration; riparian restoration projects were those projects implemented by planting of riparian vegetation or removal of nonnative vegetation as the primary or sole restoration method; watershed action projects were those in which the project was implemented up in the watershed without manipulation of the channel, and they included, for example, addition of stormwater management, creation of wetlands, or use of cover crops; and "other" projects were varied, including, for example, treatment of acid mine drainage, dam removal, changes in reservoir releases to restore natural flow regime, or creation of an in-stream or riparian wetland. (Reproduced from Palmer et al. 2014)

apparent that the most common management activities involve channels, sediments, and flows, sometimes combined as hydrogeomorphic; and addition of wood, boulders, and gravel, all measures to improve habitat for the biota. Much river restoration aims to improve the physical environment, relying on expertise in hydrology, geomorphology, and the habitat requirements of organisms.

Clearly, management activities intended to improve stream condition are many and diverse. Are they successful? If a project calls for addition of boulders and wood to a one-km river reach to improve habitat for invertebrates and fishes, the proximate measure is whether indeed the habitat elements remain in place following flood events, perhaps as observed after one or more years. But the desired outcome is self-sustaining animal populations, which may not be evaluated. The reason for emphasizing measures of success in river restoration is that much time and money has been invested, with little knowledge of what has been gained, little opportunity to learn from experience, and insufficient sharing of what works and what does not. The average costs were summed for 37,000 projects to obtain the conservative estimate that, from 1990 thru 2003, more than 1 billion dollars a year were being invested in efforts to restore US rivers (Bernhardt et al. 2005). However, only 10% mentioned any form of monitoring. A similar inventory in the UK as of 2016 contained over 2800 completed projects with only 21% stating some degree of monitoring (England et al. 2019).

When projects have been deemed successful, the criteria used may not be rigorous. Interviews with managers of over 300 US projects considered successful revealed that post-project appearance and positive public opinion were the main measures of success (Bernhardt et al. 2007). An evaluation of 44 French river restoration projects found that the quality of evaluation strategies often was inadequate for understanding the link between a project and ecological outcomes, and projects with the poorest evaluation strategies generally reached the most positive conclusions about the effects of restoration (Morandi et al. 2014). Of 848 Swiss restoration projects, success was evaluated for 232, with methods ranging from very comprehensive ecological assessments to counting the number of fish through a fish pass (Kurth and Schirmer 2014). The authors commented that comparison of results among projects and with projects elsewhere was difficult because individual projects varied in aim and method of evaluation.

The above makes clear that many restoration projects have been undertaken at considerable expense and with minimal systematic accounting. And while the intent is laudatory, efforts to re-shape rivers are significant interventions, often involving heavy equipment (Fig. 15.3). Adding to the

Fig. 15.3 A restoration crew works to add large wood to a Pacific Northwest stream where this material was aggressively removed for decades. Mckenzie River, Oregon. Photo by David Herasimtshschuk, Freshwaters Illustrated

concern, restoration projects have been observed to fail physically, or not achieve desired ecological outcomes. Explosive growth in the field of river science has drawn valid criticism for inappropriate project design (Kondolf 2006), an emphasis on restoring structure rather than function (Palmer et al. 2014), inadequate monitoring (Bernhardt et al. 2005; England et al. 2019), and lack of demonstrable improvement in the biota (Palmer et al. 2010; Feld et al. 2011). A synthesis of findings from over 800 recent European Union (EU) river restoration projects identified four ways in which projects are prone to failure: inadequate consideration of large-scale pressures related to land use in the catchment, absence of source populations to re-colonize a site, inadequate habitat and microhabitat, and failure to rejuvenate processes related to flow and sediment regime (Friberg et al. 2016).

Despite these legitimate criticisms, river restoration has helped to drive fundamental research to address knowledge gaps that limit successful restoration (Wohl et al. 2015). Individual projects have unquestionably restored some elements of river function, allowing the river's natural dynamism to reconfigure the channel and associated habitat features (Fig. 15.4). A recent evaluation of existing river restorations across Europe concluded that outcomes have been highly variable with, on balance, more positives than negatives (Friberg et al. 2016). Modest success is attributed to the local scale at which much restoration is carried out, largely ignoring the larger-scale pressures related to catchment land use or the lack of source populations to recolonize restored habitats. What one sees at the local scale in a natural river ecosystem is determined not only by micro-habitat features of the site, but by processes nested within and occurring at larger spatial scales in a hierarchy of controls. Restoration activities likewise will be scale dependent and linked to the spatial and temporal heterogeneity provided by natural stream reaches, as illustrated in Fig. 15.5. One very important implication is that efforts undertaken at the local scale may fail to produce the desired outcome, if stressors operating at a large scale remain unaddressed.

We should not under-estimate the scientific and technological knowledge needed to provide general guidance for management actions intended to restore stream ecosystems towards a good ecological status. This complexity is illustrated here by a conceptual model depicting the hierarchical relationships between catchment land-use, catchment pressures, riparian buffer management, instream abiotic states and instream biological states (Fig. 15.6). Establishment of riparian buffers has been shown to be an effective form of river restoration, acting to stabilize streambanks, mitigate diffuse pollution by agriculture, and moderate stream temperatures (Feld et al. 2011). However, assessing the outcomes from riparian planting is difficult because of the many features that characterize riparian buffers, such as buffer length, width, and density, or the species planted, as well as the multiple pathways by which buffers affect stream processes and potentially mitigate human influences. Based on a structured literature review of a large number of studies published between 1990 and 2017, Feld et al. (2018) concluded that riparian management has beneficial effects on the supply of coarse particulate organic matter, large woody debris, and shade (and thus thermal damping) that are largely independent of conditions further upstream in the catchment. In contrast, expected benefits in retention of nutrients and fine sediments from riparian management are more likely to be affected by conditions upstream of the restored section, thus requiring catchment-scale as well as local interventions. Given the substantial number of restoration activities that may be considered, the multiple response variables, and the need to take into account upstream and catchment-scale pressures, effective restoration design is a non-trivial exercise in the application of best scientific knowledge.

The shortcomings of existing river restoration practice are increasingly well understood. As Wohl et al. (2015) point out, criticisms fall into three main categories. Monitoring commonly is inadequate to quantitatively and objectively determine whether restoration goals are achieved. Many restoration projects fail to achieve significant improvements as shown by measures of water quality or biological communities. Finally, and perhaps most importantly, inclusion of the nonscientific community in river restoration planning and implementation often is inadequate. Even so, the growing practice of river restoration provides a testing ground for scientific understanding of rivers, and a context in which societal attitudes toward rivers and humanity's ability to sustain river ecosystems can advance.

Fig. 15.4 Views of the Mareta River, Italy, before (left, 2005) and after (right, 2010) river restoration that removed grade-control structures. *Source* Archivio fotografico dell'Agenzia per la Protezione civile/Luca Messina–Provincia Autonoma di Bolzano

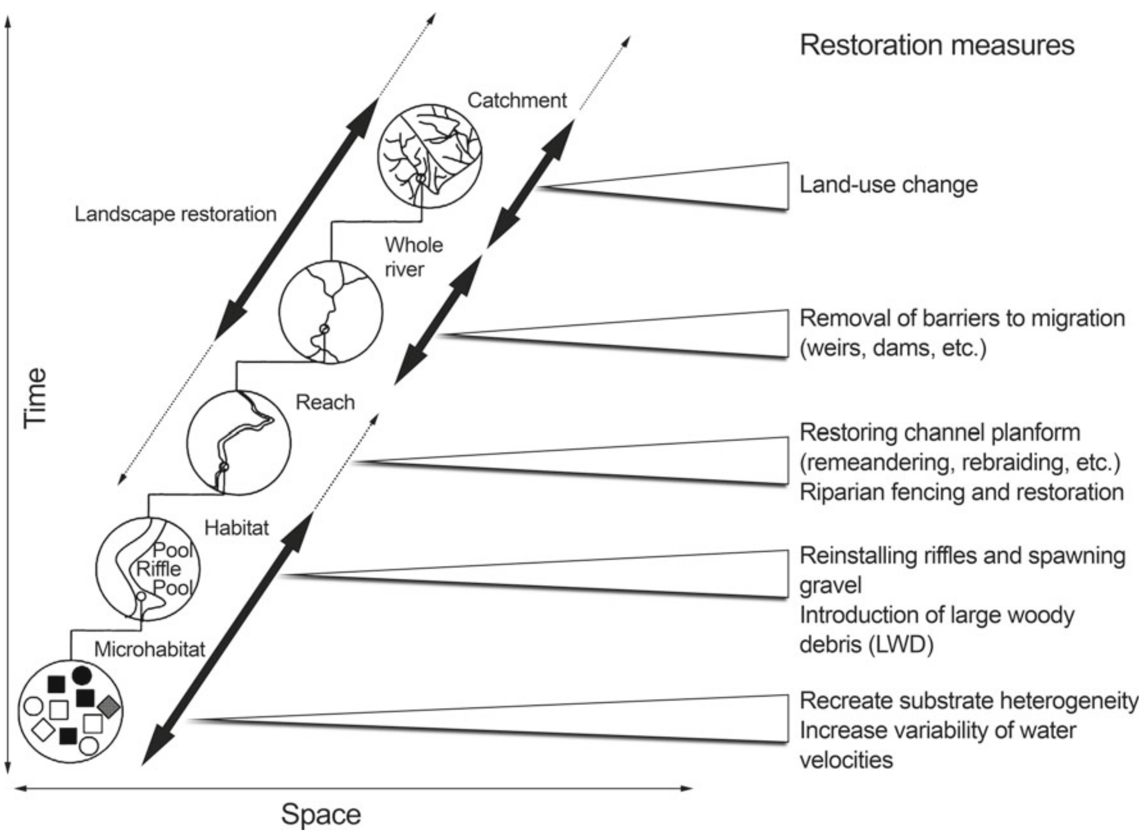

Fig. 15.5 Restoration challenges often are present across a range of spatial and temporal scales, such that impairments at the catchment scale may limit success of restoration efforts undertaken at the local scale. (Reproduced from Friberg et al. 2016)

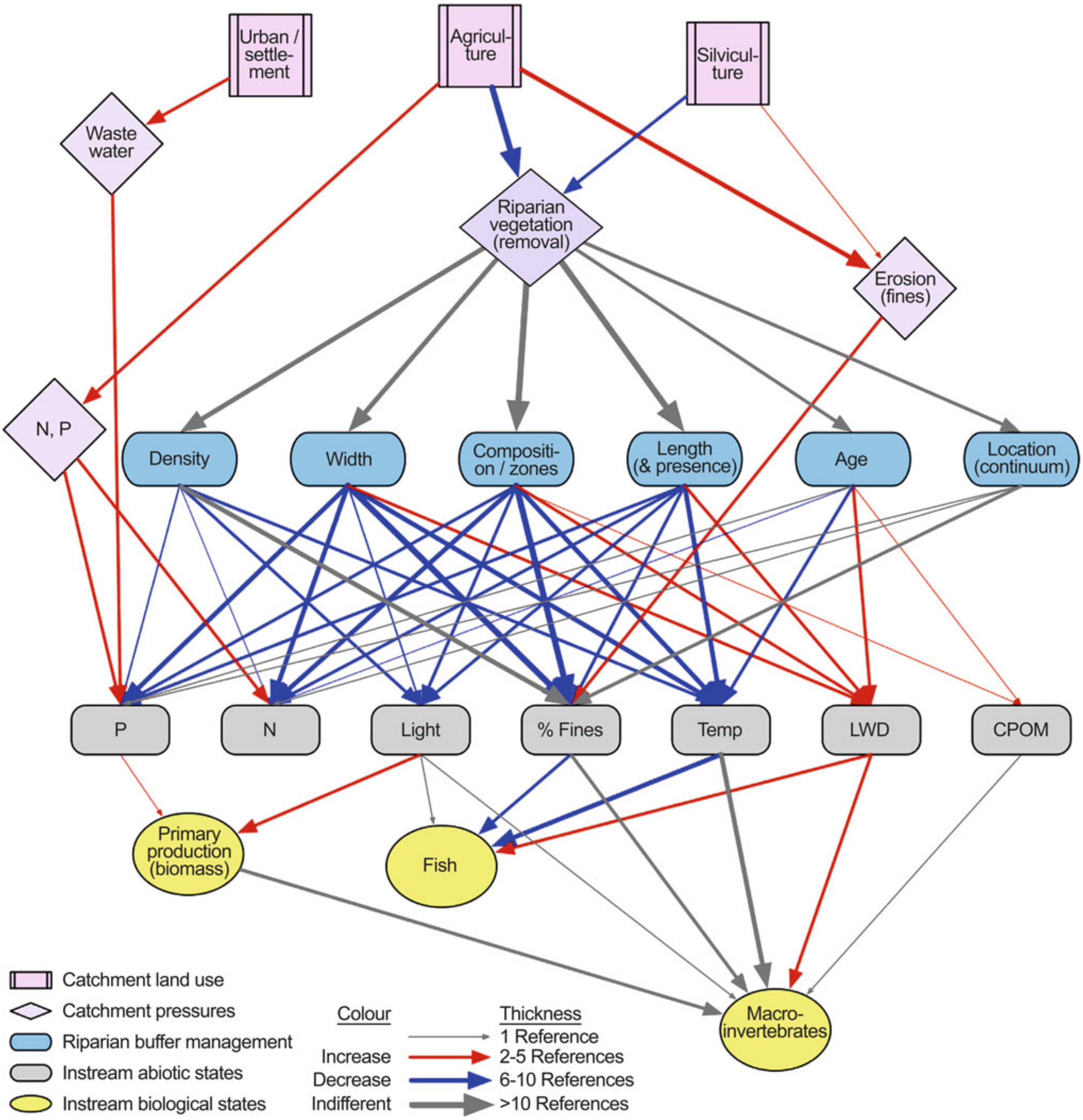

Fig. 15.6 A conceptual model based on extensive literature synthesis depicting the hierarchical relationships between catchment land-use, catchment pressures, riparian buffer management, instream abiotic states, and instream biological states. Arrows represent consistent evidence of negative (blue) and positive (red) relationships, or unclear evidence (grey) with both positive and negative effects reported in the literature. Arrow thickness is proportional to the number of studies supporting a significant relationship between two elements of the model. (Reproduced from Feld et al. 2018)

15.3 Frameworks for River Management

15.3.1 Integrated River Basin Management

Policy prescriptions for the management of water resources have long agreed on two fundamental principles. First, the river basin (catchment, watershed) is the appropriate scale for organizing water management, because water sources and uses in a watershed are interrelated. Second, because political boundaries rarely correspond with watersheds, and watershed-scale decision making structures typically are lacking (although that is changing), they should be created. Watershed-scale organizations are needed to bring together all "stakeholders" and produce integrated watershed management, avoiding a fragmented approach to management and decision-making. The authority for integrated actions likely rests with cooperative coordination among existing agencies, facilitated through some basin-wide entity that promotes a common agenda by serving as the advocate for shared priorities and an integrated approach.

Despite the attractiveness and the consistency of this message, integrated river basin management or IRBM has met with criticism for failing to achieve in practice what is intended, with shortcomings often blamed on social and political obstacles. More specifically, such efforts struggle to resolve fundamental political questions about where boundaries should be drawn, how participation should be structured, and how and to whom decision makers within a watershed are accountable ((Blomquist and Schlager 2005). Decisions typically involve trade-offs, and it is difficult to imagine all of the local impacts of different choices. Small-scale local users may feel a loss of control over resource allocation decisions as their region is subsumed under decisions made from the perspective of the larger watershed.

Integrated river basin management as a multi-faceted planning process has been re-invigorated and formalized under the Water Framework Directive (WFD) of the European Union. The Water Framework Directive, in force since 2000, is considered to be the most significant piece of European water legislation in decades, modernizing much of earlier EU water legislation and extending the concepts of river basin management to the whole of Europe (Griffiths 2002). Its aim is to take a holistic approach to water management and achieve "good water status". Key elements included water management at hydrological scales, the involvement of non-state actors in water planning, and various economic principles such as cost-benefit analyses, as well as a common strategy to support the 28 EU member states (Boeuf and Fritsch 2016). Ecological status is determined from biological parameters referenced to what would be expected in the absence of significant anthropogenic influence. The need for some exceptions is recognized; for example, for bodies of water that are artificial in construction or where the physical structure has been irrevocably and heavily modified.

The WFD has stimulated an enormous amount of activity across the European Union, leading to numerous project-specific publications and several reviews of progress (Hering et al. 2010; Boeuf and Fritsch 2016; Voulvoulis et al. 2017). Perhaps unsurprisingly, given the ambitious intent of the WFD, difficulties with implementation have received a good deal of attention, and attainment of good status for many waters remains to be demonstrated. Among the challenges, Boeuf and Fritsch (2016) point to the mismatch between ecological (river basins) and political (political and administrative institutions) scales, lack of attention to synergetic ecological effects, and low acceptance by target groups.

As river restoration matures both as science and practice, there is increasing recognition of the need for a planning framework to guide practitioners and place project-specific restoration within a river basin context. A critical review of 663 published studies of European restoration projects identified poor or improper project planning as the most frequent shortcoming (Angelopoulos et al. 2017). One recently proposed planning framework is depicted in Fig. 15.7a. At the project identification stage, clear objectives are set for ecological condition at local scale, while keeping the project in a river basin/catchment context. Benchmarks are measurable targets for restoring degraded river sections by comparison with sites that have the required ecological status in that river system or elsewhere. Endpoints are feasible targets for river restoration and so are the basis for determining success or failure. Monitoring can include a wide range of physical, chemical, and biological variables, depending on what concerns motivated the project, but should relate to outcomes. In other words, if habitat elements are added to benefit fish populations, it is useful to quantify the habitat created, but important to assess whether fish populations benefited. Monitoring should include before and after sampling, and will be even more insightful if paired with a control site not receiving restorative measures. A full evaluation of project success requires clear objectives, endpoints, and measurable indicators that are sensitive to gradual improvements (Fig. 15.7b).

Regardless of mixed progress to date, the WFD has accomplished a great deal. Implementation of the WFD is greatly increasing knowledge on the ecology of European surface waters. Rather than relying mainly on chemical quality of surface waters, condition is assessed using a wide range of biological measures referenced against best

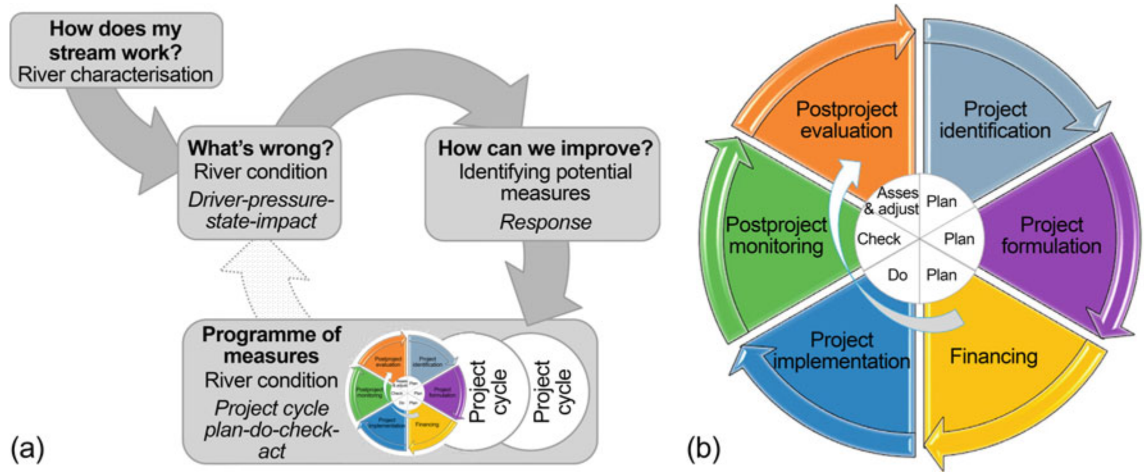

Fig. 15.7 (**a**) Project planning cycle at a catchment scale using a five step approach starting at the top left text box: (1) River characterization—at a catchment scale to identify river styles and understand their processes. (2) River status—understand the current condition of the aquatic biota or biological quality elements. (3) River restoration potential—to understand the level of restoration a river can achieve. (4) Project identification—to identify specific restoration projects at a reach scale and identify suitable rehabilitation measures and project objectives. (5) The project cycle—planning, formulation and implement of projects at a local scale. (**b**) Six stages for restoration project planning: (1) project formulation; (2) financing; (3) project implementation; (4) post-project monitoring; (5) post-project evaluation and (6) adjustment or maintenance. (Reproduced from Friberg et al. 2016)

attainable condition for a water body type, thereby placing aquatic ecology in the center of water management. Monitoring methods are being improved and standardized. Much attention is given to difficulties encountered in the planning process cycle, and with engagement of the public, areas where further improvements are needed. Mechanisms are in place for river basin management across national borders, including international commissions for transboundary basins such as the Rhine and the Danube. Much can be learned from this ambitious experiment to improve water quality and ecological status throughout the European Union, which cannot be recounted briefly. Interested readers may consult EEA (2018) and Carvalho et al. (2019).

15.3.2 The US Clean Water Act and TMDLs

Protection of the waters of the United States is largely accomplished through the Clean Water Act (CWA), which provides the basic structure for regulating discharges of pollutants, giving the Environmental Protection Agency (EPA) authority to implement pollution controls. Other Acts of Congress, including the Endangered Species Act, National Environmental Policy Act, Wild and Scenic Rivers Acts, and Surface Mining control and Reclamation Act, among others, provide further means to protect freshwater ecosystems. Numerous federal and state agencies have roles in making decisions and implementing regulations, and nongovernmental organizations (NGOs) such as The Nature Conservancy, the Sierra Club, and American Rivers can bring attention and pressure when regulatory enforcement is perceived as lax. However, it is the CWA, and its main regulatory tool, the total maximum daily load (TMDL), that is the principal mechanism for managing freshwater ecosystems. Many of the US restoration projects described earlier likely were initiated in response to a TMDL finding.

The objective of the Clean Water Act (CWA) of the United States, in effect since 1972, is to restore and maintain the chemical, physical, and biological integrity of the nation's waters. It requires states to compile lists of water bodies that do not fully support beneficial uses such as aquatic life, fisheries, drinking water, recreation, industry, or agriculture. These inventories are known as 303(d) Lists, and characterize waters as fully supporting, impaired, or in some cases, threatened for beneficial uses. Water quality standards set by a state, territory, or authorized tribe provide a narrative and numeric criteria for determining whether a waterbody is attaining or not attaining its designated uses; waters designated as not attaining then require the establishment of total maximum daily loads (TMDLs) for all pollutants identified as causing impairment. The US EPA assists states (this term includes territories and authorized tribes) in listing impaired waters and developing TMDLs for these waterbodies (Fig. 15.8). A TMDL is the maximum amount of a pollutant allowed to enter a waterbody so that the waterbody will meet and continue to meet water quality standards for that particular pollutant. It is determined by the sum of all point and nonpoint sources entering the waterbody, plus a margin of safety. The EPA's regulations require public involvement in developing TMDLs, although the level of citizen involvement varies by state. Once completed, TMDLs should clearly identify the links between the waterbody use impairment, the causes of impairment, and the pollutant load reductions needed to meet the applicable water quality standards. States are not explicitly required to develop TMDL implementation plans, although many include some type of implementation plan with the TMDL. The TMDL alone is sufficient to remove the waterbody from the state's list of impaired waters.

Management of point-source pollutants generally is implemented through a permitting process under another section of the CWA. Reductions in non-point sources are implemented through a wide variety of regulatory and voluntary programs, and incentivized by EPA program funds that grant money to the states to fund specific projects aimed at reducing the nonpoint source pollution. Substantial progress has been made in improving water quality through regulatory and permitting processes for wastewater treatment plants and industrial dischargers, two prominent point

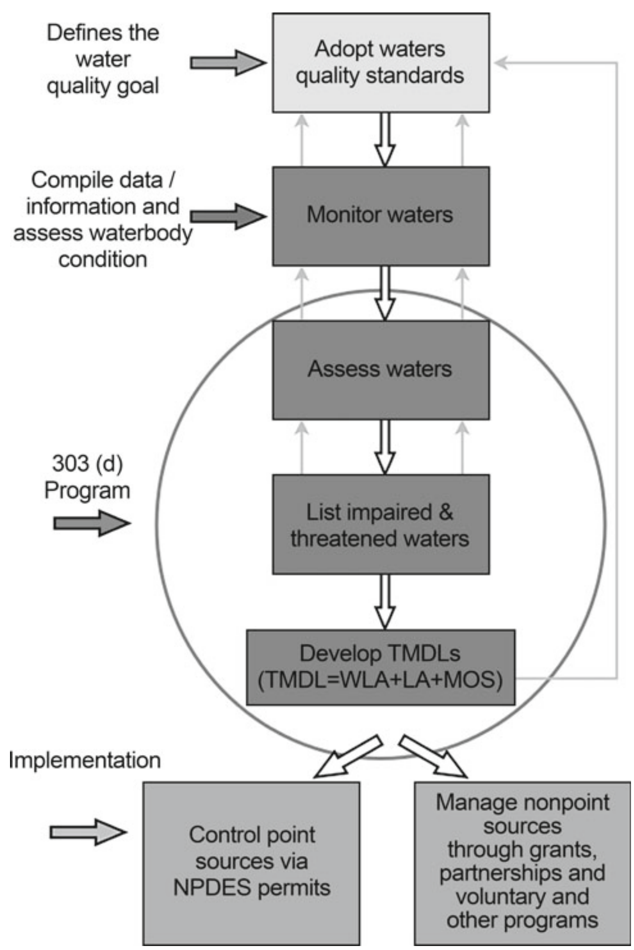

Fig. 15.8 A depiction of the TMDL process under the Clean Water Act. WLA, wasteload allocation; LA, load allocation; MOS, margin of safety (Reproduced from https://www.epa.gov/tmdl)

sources (NRC (National Research Council) 2001). However, control of unregulated nonpoint sources of pollution has been less successful, and largely for that reason, the nation's water quality goals of "fishable and swimmable" have not been achieved.

The TMDL program has been controversial, in part because of requirements and costs faced by states, as well as by industries, farmers, and others who may be required to use new pollution controls to meet TMDL requirements. Often, prodding by lawsuits brought by NGOs or citizen groups is necessary to move the process along. Development of a TMDL does not guarantee that improvements will follow. Success at achieving targets is considerably higher when addressing point sources, which are managed through a permitting process, than with nonpoint source pollution, which usually requires the coordination of a suite of voluntary activities. Most TMDLs have a non-point source component, and common barriers to success include inadequate funding, incomplete knowledge of the effectiveness of various management practices such as riparian buffer strips, cover crops, and other land management measures, and inability to demonstrate causal connections is system response.

The TMDL process increasingly is being used to address water quality impairments in large systems where nonpoint pollution is the main driver of impairment and the most effective combination of best management practices (BMPs) and their spatial deployment is largely uncharted territory. BMPs typically are intended to retain stormwater and enhance infiltration so that nutrient processing and sediment storage can reduce loads to downstream systems. Their efficiencies vary and depend on design, maintenance and placement within the watershed.

The TMDL process has been used to implement BMPs over a large region in the case of the Chesapeake Bay of the eastern US. The Bay TMDL (www.epa.gov/chesapeake-bay-tmdl), completed in 2010, identified reductions in total nitrogen, phosphorus, and suspended solids needed to meet water quality standards, and specified that 60% of its goals be implemented in 2017, with total implementation by 2025. The need to reduce nutrient and sediment loads to meet these requirements has led to the implementation of novel stream corridor restoration designs, including the conversion of eroded channels to stream—wetland complexes that enhance a channel's capacity to trap and retain suspended materials delivered from upstream. Unfortunately, their effectiveness has been less than desired. Input-output budgets of total suspended solids at Bay tributaries where stream-wetland complexes had been constructed showed insignificant changes (Filoso et al. 2015). Rather than attempt to trap nutrients near the stream's juncture with the Bay, this research suggested that BMPs might better be placed in upstream locations where most inputs originate. Williams et al. (2017) also reported better nutrient and sediment retention from a headwater stormwater retention project than a downstream constructed wetland. Based on their findings, the authors argue that headwater restoration projects and urban BMPs are likely to be better investments than large-scale stream-wetland complexes constructed in the lower watershed.

The development of Total Maximum Daily Loads (TMDLs) has increased markedly in recent years and as of spring 2014, over 70,000 TMDLs had been completed (https://www.epa.gov/tmdl/impaired-waters-restoration-process). As staggering as this number is, the additional need is huge. The most recently reported National Rivers and Stream Assessment (USEPA 2016) gives a rating of poor to 46% of assessed stream based on biological condition. An additional 25% were rated as fair, and only 28% were found in good condition. Nutrients, riparian condition, and sediments were identified as leading causes. Implementation of TMDLs takes time, and demonstration of success likely requires even more time, making it difficult to quantify. Published research on TMDL projects does not appear to be common in the scientific literature, and broad, comparative assessments appear to be lacking. There likely is overlap between projects initiated in response to a TMDL and many of the restoration projects described earlier, where success often is not adequately assessed. The fact that implementation of TMDLs is not required for de-listing adds another layer of uncertainty in evaluating success of this program.

15.3.3 Freshwater Conservation Planning

The rationale for conservation planning is clear: to provide adequate protection to the full complement of species and ecosystem types in freshwaters. Surveys find that as much as one in three freshwater species are threatened with extinction world-wide, as described earlier in this chapter (Fig. 15.1), and freshwater taxa often exhibit a higher risk of extinction than their terrestrial counterparts (Collen et al. 2014). Regrettably, freshwater conservation planning has lagged behind terrestrial and marine efforts. Despite increasing efforts to establish protected areas (PAs), their effectiveness for freshwater conservation is uncertain and freshwater biodiversity continues to decline (Hermoso et al. 2016). Among the principal reasons are lack of consideration of freshwater needs when designing protected areas, fewer resources devoted to freshwater conservation management, and poor understanding of complex management problems beyond the limits of the protected area. Limited information on the geographic distribution of species and ecosystem types has also hampered freshwater conservation planning. Fortunately, however, efforts to develop global syntheses of freshwater biogeography and threats are making good progress (Abell et al. 2008; Vörösmarty et al. 2010). A map of freshwater ecoregions of the world, based on the distribution and composition of freshwater fish species, provides a useful tool to support global and regional conservation planning efforts, and to identify outstanding and imperiled freshwater systems. Presently including 830 ecoregional units, an interactive map can be viewed at https://www.feow.org/. Combining high-resolution hydrographic and land-use data, Grill et al. (2019) have generated a global map of free-flowing rivers, thereby identifying least-impacted areas that can be viewed as conservation opportunities.

At this time, freshwater protection relies heavily on protected areas designed largely around terrestrial features, with often limited consideration of their effectiveness in representing freshwater features of conservation concern (Hermoso et al. 2016). Rivers located within parks have experienced contaminant spills and invasive species, and often are affected by dams even within park boundaries. Most protected areas are not designed with biodiversity protection as a goal, and so whether their boundaries include species of concern may be accidental. Existing freshwater protected areas often are situated downstream from disturbed lands (Abell et al. 2007), in some cases rivers form the border of a reserve and so receive protection on only one side (Roux et al. 2008), and many are small, fragmented areas that lack sufficient connectivity to a broad suite of habitats (Pringle 2001). In France, all mainland national parks are located at high elevations, whereas most imperiled fishes occur downstream (Keith 2000). Using a database of conservation and recreational lands in the state of Michigan, US, Herbert et al. (2010) found uneven representation of key freshwater features. Wetlands were well represented, but riparian zones were not, particularly for headwater streams and large rivers, and terrestrial rare species received better coverage than their aquatic counterparts.

At present, nearly 15% of the world's land is in some form of protected area, close to the goal of the Convention on Biological Diversity to conserve 17% of inland waters by 2020 (CBD 2010). (Visit the CBD website for most recent statements of goals and progress at https://www.cbd.int/). For freshwater conservation to be effective, however, we require both better knowledge of the diversity and distribution of taxa and ecosystem types, and conservation planning based on an understanding of freshwater ecosystems.

When knowledge of the diversity and distribution of many taxa is inadequate, it often is assumed that better-known groups will act as surrogates for conservation planning purposes (Rodrigues and Brooks 2007). Yet, comparisons of the geographic distributions of terrestrial and aquatic taxa generally find that the former are inadequate surrogates for patterns of both richness and threat for many freshwater groups. Utilizing a comprehensive assessment of freshwater biodiversity for the entire continent of Africa, Darwall et al. (2011) examined patterns of richness and threat for all known species of freshwater fish, crabs, mollusks, dragonflies, and damselflies, and compared patterns for these aquatic groups with those of birds, mammals, and amphibians. In general, they found that groups that have been the focus of most conservation research are poor surrogates for patterns of both richness and threat for many freshwater groups, and the existing protected area network underrepresented freshwater species. In addition, freshwater groups had significantly lower surrogacy values for each other than did birds, mammals, and amphibians. In short, conservation efforts targeted at the better-known taxonomic groups may not confer adequate benefits for other species. In their global survey six freshwater taxonomic groups, Collen et al. (2014) also found little congruence among these groups in species richness, threatened-species richness, and endemism.

There also is recognition that, due to their linear, branching, and hierarchical nature, aquatic systems are not well suited to terrestrial PA planning approaches. Authors agree that the catchment scale is appropriate for freshwater conservation (Saunders et al. 2002; Dudgeon et al. 2006), but problematic in practice because the area required can be impractically large and the exclusion of people rarely is feasible. When one considers the need to protect the entire upstream drainage network, the riparian zone, and much of the surrounding landscape, and to avoid dams, pollution, or other activities that might prevent passage of migratory species, the challenges of whole-catchment conservation are apparent. Abell et al. (2007) argue that the solution requires looking beyond the protection of individual sites, and instead developing a spatially distributed set of conservation strategies intended to protect specific populations or target areas.

The literature addressing effective design of freshwater PAs has largely combined core principles of freshwater ecology with common sense (Hermoso et al. 2016). These principles emphasize the importance of preserving upstream–downstream and lateral linkages both for biophysical functioning and species movements; that abating threats requires a catchment-wide approach, and that maintaining or restoring hydrological regimes is critical to a PA's success. Further, PAs should be representative of different types of freshwater ecosystems, large enough to provide adequate habitat, sufficiently connected via upstream-downstream linkages to allow the movements of biota and transport of materials, and cognizant of external threats. Various approaches to freshwater conservation planning have been explored, including manual exercises and others automated by software, but the end product is the same: a map of locations targeted for conservation action, superimposed on a river network, showing levels of human disturbance, extent of protection, and gaps in the protected network. From this one can inventory the number of river segments by level of disturbance, level of protection, and priority for conservation action.

Development of a network of PAs for the Upper Mississippi River basin illustrates a sequential process that combines a coarse-and fine-filter approach. The coarse filter relies on a hierarchical spatial classification based on broader scale zoogeographic and hydrologic units, and the fine filter uses detailed species distribution data where available (Higgins et al. 2005). Planning for the Upper Mississippi River basin benefited from a relatively large amount of data on species occurrences available for the fine filter (Khoury et al. 2011). The coarse filter identified 238 unique types of aquatic ecosystems which, combined with the 129 species that were elements of the fine filter, resulted in 606 areas of biodiversity significance, primarily small rivers, headwaters, and creeks (Fig. 15.9). If implemented, this network would ensure some representation for 78% of the 129 fine-filter species. Working in a data-scarce region where biological and physical data were almost completely lacking, Thieme et al. (2007) used remote sensing to map basins/sub-basins for a large river system in the southwest Amazon and classify ecosystem types based on physical features and vegetation. The resulting network of conservation areas was

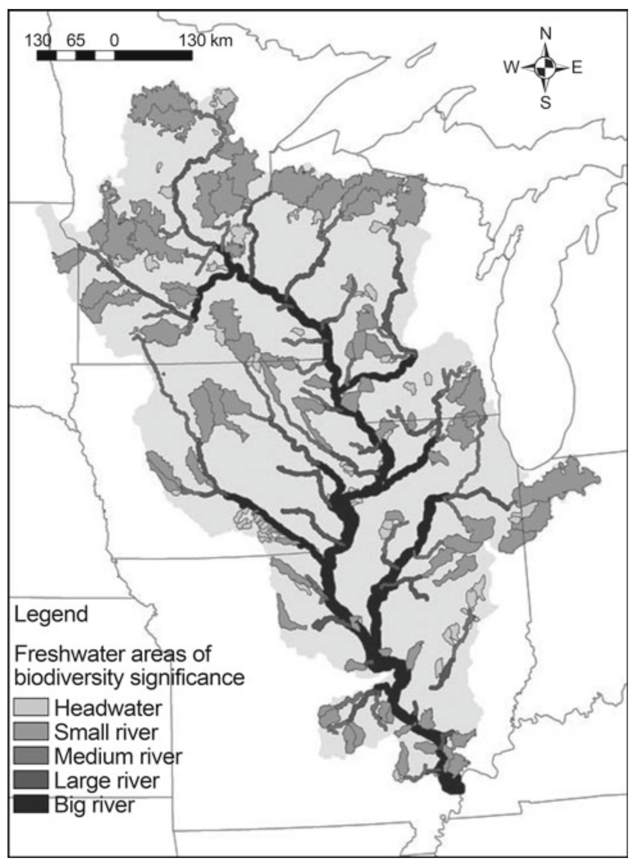

Fig. 15.9 Freshwater areas of biodiversity significance in the Upper Mississippi River. (Reproduced from Khoury et al. 2011)

partially encompassed within protected lands and indigenous areas, but their analysis identified 84 currently unprotected sub-basins necessary to fulfill representation and connectivity goals.

Further evolution of freshwater PA planning lies in the direction of formalizing the necessary steps, and increasingly is computationally intensive, often relying on specialized software to establish a network of protected areas representing the full variety of species or ecosystems. Referred to as systematic conservation planning, it emphasizes three overarching principles—representation, persistence, and quantitative conservation target setting (Nel et al. 2009). Representation refers to the need to include the full variety of biodiversity and ecosystem types in the planning region. Persistence requires maintenance of ecological condition to support the natural processes that maintain species and ecosystem integrity; in effect, assessing and addressing threats. Setting quantitative conservation targets requires defining the number of occurrences of a particular river type or the number of occurrences of a species that are desired. Together these allow computation of various conservation portfolios by calculating the contribution each site makes to conservation targets not yet achieved in the existing set of conservation areas. Many other factors must be taken into consideration, of course. Vulnerability of a site, essentially a threat assessment, can be assessed using information on planned or potential development within a planning unit. A number of societal considerations are critical to project implementation and success, including cooperation among different actors, building capacity in conservation agencies, raising awareness of the need for conservation, and developing an appropriate monitoring and evaluation system.

An advantage of computer-driven planning algorithms is the ability to examine alternative scenarios, vary conservation targets, and factor in costs. Esselman and Allan (2011) used Marxan conservation planning software to design a network of river sites intended to capture 15% of the range of each of 63 fish species in Belize, Central America. Upstream risk intensity was modeled from a GIS of landscape-based sources of stress, and solutions were constrained to account for river basin divides. The proposed reserve network encompassed 11% of the study area, of which half was within existing protected areas, and remaining areas were identified as gaps in protection. Addition of critical management zones, as defined by Abell et al. (2007), including riparian buffers and fish migration corridors, expanded the network area by one-fifth.

The principles of freshwater conservation planning are increasingly understood and embraced by agencies and NGOs. However, many completed plans are research undertakings whose impact on actions in the real world is not obvious. Some, such as the Upper Mississippi Basin plan, are the work of influential NGOs such as The Nature Conservancy, with presumably a higher likelihood of implementation. Expanded monitoring and evaluation of freshwater PAs is of critical importance to learn what influences their effectiveness and, hopefully, to demonstrate their benefits. As with restoration, implementation, monitoring, and assessment of effectiveness are challenges still to be met.

15.4 Three Pillars of River Management

Successful management actions to repair, restore, and protect rivers will require expertise from many sectors, confidence that actions taken are likely to produce desired results, and support from the public and institutions. Here we put forth what might be called the three pillars of river management: fundamental science, measurement of progress, and societal support.

15.4.1 Understanding the Fundamentals of River Systems

Current scientific knowledge provides a sound underpinning for river management. Recent years have seen significant advances in methods of hydrologic analysis, flow metrics that have ecological relevance, and our understanding of the relationships between flows and ecological processes. Using data on the relationship between flow and ecology over a wide range of flows and species, including life cycle stages and seasonal timing, the science of environmental flows, or e-flows, offers a holistic approach with the potential to recommend a hydrologic regime that can achieve desired outcomes linked to explicit quantitative or qualitative ecological, geomorphological, and perhaps also social and economic responses. The ecological limits of hydrologic analysis (ELOHA) provides a framework for determining environmental flows needed to meet ecological and societal needs (Fig. 2.20). Beginning with hydrologic analysis and streamflow classification, this multi-step framework assigns flow-altered streams to a presumed pre-impact stream type, using flow alteration-ecological response relationships drawn from existing data and knowledge or new studies. More than a science framework, ELOHA seeks to incorporate expert and traditional knowledge and differing priorities and social perspectives to provide a decision-making framework to aid planning and address water conflicts.

Research in fluvial geomorphology (Chap. 3) provides a process-based understanding of the balance between river discharge and sediment supply, how together they determine many aspects of channel form and habitat, and how alterations to either can be major drivers of river degradation. This is the knowledge base to understand the influence of altered flows and changes in sediment supply, especially reductions in sediment supply due to trapping behind dams and mining of river beds for gravel. Stream restoration requires an understanding of the interactions of discharge and sediment supply in determining channel form and physical habitat within the channel. Most restoration work begins with these considerations, often referred to as hydrogeomorphic because of the intersection of hydrologic and geomorphologic processes. Recognition of the dynamic nature of river systems allows us to make realistic choices between when to do nothing, when to undertake management interventions to assist system recovery, and when the best solution is engineered design, necessary to accommodate human presence in the landscape (Fig. 15.10).

How organisms respond to different features of their abiotic environment, including dissolved oxygen, current, temperature, and physical habitat elements, has received extensive study for decades (Chap. 5). The roles played by variability of flows, substrate heterogeneity, wood, alternating pools and riffles, gravel for spawning by fishes, habitat for invertebrates, and surfaces for attachment by algae and biofilms, are well understood. It has long been recognized that structurally simple or extreme environments tend to support fewer species, whereas more moderate and heterogeneous habitats support more species. Thus, the knowledge exists to reverse the anthropogenic degradation and homogenization of habitat resulting from human disturbance.

To be sure, our knowledge is incomplete, and complex interactions among the main drivers of environmental degradation make it difficult to predict with certainty how a system will respond to management intervention. The conceptual model of linked pressures and response variables associated with riparian buffer management (Fig. 15.6) illustrates the complexity of our current understanding of cause-and-effect relationships, and the amount of research examining each linkage. Expectations of outcomes from management intervention must always be paired with acceptance of uncertainty, and thus the need for evaluating system response and possibly for the employment of new management measures.

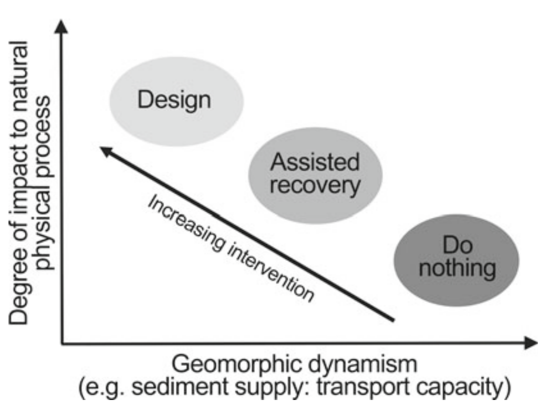

Fig. 15.10 The need for restoration intervention is a function of degree of impact and geomorphic dynamism of the system at relevant scales. (Reproduced from Friberg et al. 2016)

15.4.2 Measuring Progress

Monitoring and assessment are key aspects of any management program seeking to improve ecosystem condition. Ideally, this takes place within a larger framework in which objectives are clearly established, management is targeted at probable causal factors, and lessons learned from a rigorous monitoring program feed back into future actions in a cycle of adaptive management. Monitoring of water quality and sensitive species, especially of species responsive to organic enrichment and low oxygen levels, has been practiced for at least a century. Methods began to change after the 1970s, at least partly because 1972 amendments to the Clean Water Act called for maintaining and restoring the biological integrity of fresh waters. Today, monitoring to assess river condition is increasingly sophisticated and standardized, employing integrative ecological indices based on the biota and on aspects of habitat. The goal of these indices is to measure river condition, and increasingly this is referred to as 'river health', in the very broad sense that a healthy river is one in good condition (Karr and Chu 1999).

Development of the Index of Biological Integrity (IBI, Karr et al. 1986), was a significant milestone, as it provided the link between the goal of maintaining and restoring the biological integrity of fresh waters, and the means to measure its attainment. The IBI is a multi-metric index, meaning that it is the sum of ten or more individual metrics, including species richness and composition, local indicator species, trophic composition, fish abundance, and fish condition. Because it is based on multiple metrics, which are expected to be sensitive to different levels and types of environmental stress, the IBI is considered a useful integrator of multiple stressors affecting biological assemblages. By the late 1990s, a multitude of rapid bioassessment protocols were in use by various state agencies in the US (Barbour et al. 1999), with subsequent extensive updates (USEPA 2013, 2016). This provided technical guidance to protocols, although the choice of metrics often differed among states. In Europe, implementation of the WFD has led to the development of many different approaches, which, although diverse, are sufficiently intercalibrated to provide robust comparisons across countries and regions (Birk et al. 2012). Multimetric indices based on benthic invertebrates are most common, but other methods, including those based on traits, biomarkers, and functional feeding groups, also are used (Bonada et al. 2006).

The need for indices suitable for monitoring river condition based on the biota has led to much innovation and a diversity of approaches. A comparison of 13 protocols for monitoring streams from different regions and countries found that methods of sampling and level of taxonomic resolution were similar (Buss et al. 2015). However, translation of data into an index has followed different paths, resulting in at least three main approaches. Multi-metric indices, described above, are in wide use. In addition, biotic indices based on the presumed sensitivity of individual taxa to impairment assign a sensitivity or tolerance score to each family or genus of aquatic invertebrates, and then aggregate these into a single score based on the assemblage of invertebrates at a site. The Saprobian system, long used in Europe to assess organic pollution (Bonada et al. 2006), and the Family Biotic Index (Hilsenhoff 1988), developed in North America, are examples. "Predictive modeling" is another approach that uses statistical models to predict the expected set of species from environmental site characteristics based on a multivariate model developed using undisturbed reference sites. When a test site is to be evaluated, its environmental conditions are used to predict the expected assemblage assuming the site is unaltered, and then the observed assemblage is compared to the expected (O/E). Predictive modeling has been pursued in England (Clarke et al. 2003), Australia (Simpson and Norris 2000), and the US (Hawkins et al. 2000). The US National Rivers and Streams Assessment uses both multi-metric and O/E indices for assessing wadeable streams (USEPA 2016). In the future, methods in the early stages of development, including genetic methods that rely on DNA sampled from organisms or directly from the water ("environmental DNA") may play a more prominent role (Hering et al. 2018).

Each approach requires the establishment of reference conditions for comparison with test or impaired sites. While the need for benchmark or reference conditions is widely recognized, locating a suite of undisturbed sites often is challenging. Stoddard et al. (2006) advocate use of terms such as minimally disturbed and least disturbed to acknowledge that reference condition may not reflect the historical, undisturbed ecological condition of streams. Use of reference condition encounters another challenge when bioassessment covers large geographic areas with differing climate, geology, vegetation, etc. Biological assessment is most efficient when reference condition is regionalized, and so comparisons across Europe or the United States rely on indices that are referenced to different regional expectations. To aid in implementation of the WFD, European workers have developed a river typology based on altitude, size, and geology, which captures almost 80% of all European rivers for purposes of intercalibration of monitoring results (Lyche Solheim et al. 2019).

The development of biological indices spurred the transition away from reliance mainly on chemical water quality standards, such as dissolved oxygen and nutrient concentrations, by providing a mechanism to determine the ecological condition of a waterbody. Indices translate a matrix of species names and abundances decipherable only by another biologist into a single metric that can be used to classify a water body as poor, fair, good, or excellent. They are a powerful tool for communicating the ecological condition of a region's or a nation's streams and rivers to decision-makers and the public. By combining sampled locations from a national assessment of US streams rated in good, fair, or poor biological condition, with a model relating stream and landscape condition, Hill et al. (2017) mapped the condition of every stream in the US. Such comprehensive information is extremely useful in identifying priorities for conservation and restoration.

A freshwater monitoring program in South East Queensland, Australia, illustrates how such information can be used to garner public support and influence decision-makers. The Queensland project took an important step beyond monitoring by converting its findings into scorecards that communicate easily and effectively with the public. Each indicator was standardized from 0 to 1, where 1 is the reference condition for a particular stream type (i.e. 'best case'), and 0 is the 90th percentile recorded or the theoretical minimum (i.e. 'worst case'). These report cards (Fig. 15.11) are well publicized, and presented each year to politicians and senior policy makers in a televised ceremony (Bunn et al. 2010).

15.4.3 Societal Support

Existing scientific knowledge and practical expertise are not the barriers to repairing, restoring, and protecting river ecosystems. The necessary technical expertise to improve the ecological condition of rivers exists. Expertise in hydrology, geomorphology, and ecology is sufficiently advanced to give confidence that the underlying principles of river science are well understood. The science and practice of monitoring ecological condition also is on sound footing, ensuring that measurement of condition and tracking of trends can be reliably accomplished. Practitioners of river restoration in the public and private sectors have a wealth of experience to draw upon, although some controversies exist regarding approaches, and whether the sharing of lessons learned is as effective as it could be. With the caveat that the response of any complex ecosystem to management intervention includes a measure of uncertainty, hence compels an adaptive management perspective, the necessary science and technical expertise is sufficient to undertake a wide range of actions to rehabilitate and restore river ecosystems.

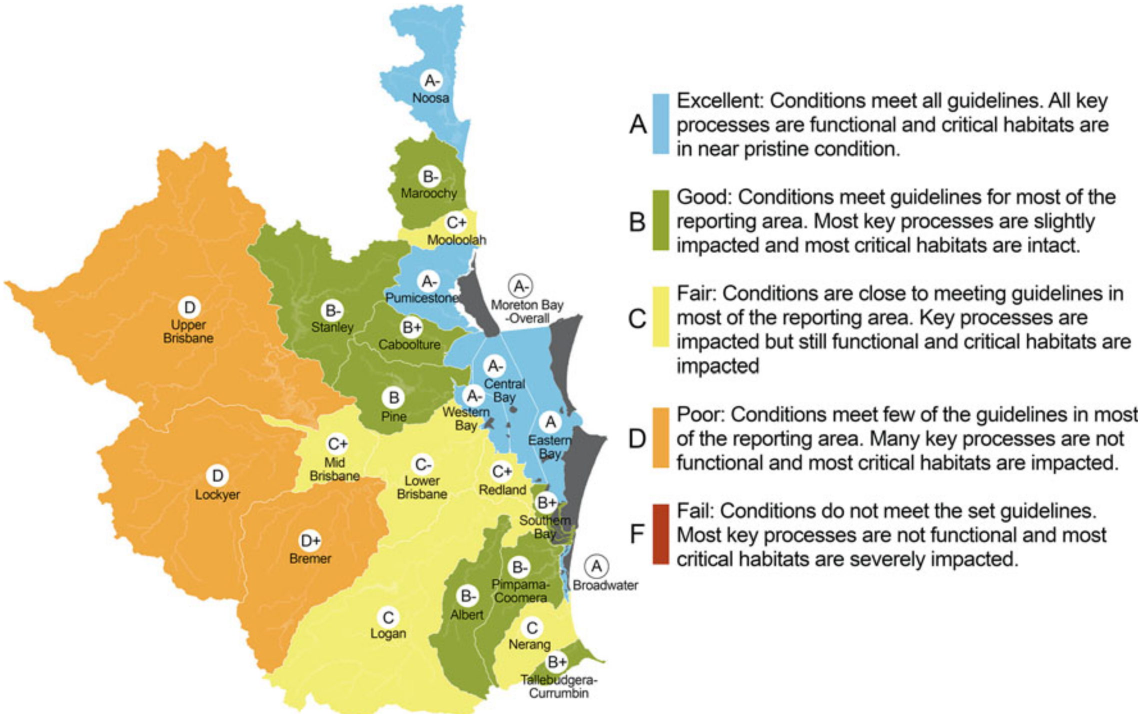

Fig. 15.11 A report card showing environmental conditions across multiple catchments in South East Queensland, Australia. This reporting system has been in place since 2000, and the report itself is presented annually by the non-governmental organization, Healthy Land and Water, to politicians and senior policy makers in a public, televised ceremony. Image courtesy of the Healthy Land and Water, Queensland, Australia

Successful river restoration requires more than this, of course. It requires the commitment of resources, both human and monetary, and the support of responsible agencies and institutions. This in turn requires more effective transmission of knowledge to policy and decision-makers, and better communication with the public. Going further, it requires recognition that ecosystems are complex, interactive socio-ecological systems (Young et al. 2006). At the same time that human economic uses of rivers and their landscapes are ultimate drivers of the many direct causes of river degradation, society desires that river resources be conserved and restored to functional states. Therein lies the great challenge—how to use riverine resources in a manner that is socially and ecologically acceptable (Naiman 2013). To fully appreciate river ecosystems as complex socio-ecological systems, and to manage them to sustainably meet societal and ecological goals, will require new approaches and further maturation. Encouragingly, however, efforts to communicate with, learn from, and actively engage the public are increasingly woven into existing efforts through the participation of stakeholders.

Stakeholders include all who have some association with the river ecosystem and may be affected by management actions. This may include scientists, representatives of interest groups and businesses, members of the community, employees of government agencies and non-governmental organizations (NGOs), and activists, representing a diversity of perspectives. Where their participation is embraced, stakeholders play an increasingly important role in public policy and are frequently used as a source to better inform public decision making. River restoration centers and environmental NGOs can have the resources to exert a strong role in shaping dialogues and influencing outcomes. Community groups can participate in the collection of data and monitoring of projects within a citizen science framework, providing the opportunity for much more extensive post-project appraisal and a more productive relationship between scientists and nonscientists. The involvement of local communities in monitoring has the potential to stimulate learning, promote engagement, and provide useful information to inform management.

Stakeholder participation goes beyond simply receiving input from the public, requiring that stakeholders have some influence over decisions made and actions taken. It also brings a mix of risks and advantages (Luyet et al. 2012). For example, stakeholder participation builds trust and local knowledge can improve project design; however, the process can be time-consuming, expensive, and not fully representative of all interests. Who participates, how they participate, and when within the project timeline they participate, are all important concerns addressed by a social science literature that discusses participation techniques. Differences in interests and objectives can complicate decision-making. One can imagine a situation where local stakeholders prioritize aesthetic goals such as a riverside path and an uncluttered river by the removal of wood, while science professionals advocate for an undisturbed riparian and in-channel wood. In a survey of local stakeholders with respect to the construction of a dam on a Dutch river, Verbrugge et al. (2017) found that local residents, recreational users, and shipping professionals differed in their level of trust, attachment to the river landscape, and evaluation of the effects of dams.

Involving stakeholders in scenario planning can be a powerful tool for envisioning alternative outcomes of river management. Based on detailed input from local stakeholders, Baker et al. (2004) developed three alternative future landscapes for the Willamette River Basin in the year 2050, and evaluated the likely effects of these landscape changes on four endpoints: water availability, Willamette River physical and biological condition, ecological condition of tributary streams, and terrestrial wildlife. Stakeholder input was extensive, with monthly meeting held for over two years to develop very detailed assumptions in designing each alternative future (Hulse et al. 2004). The process led to greater stakeholder understanding and a feeling of ownership in the final product, and strongly reflected stakeholder values, assumptions, and visions. Researchers concluded that expert-based scenarios outside the experience of current stakeholder experience may be less readily accepted, but perhaps should be blended with stakeholder-based perspectives to broaden the range of envisioned futures.

Equitably engaging stakeholders in the governance of water resources is challenging (Butler and Adamowski 2015). If not actively engaged in advocacy groups, community members may not be aware of their legal rights or their ability to have their voice heard in watershed planning activities. Historical relationships between decision-making bodies and groups of varying demographics may influence the power and perceived legitimacy of stakeholder groups. The ability to attend meetings and other events may be restricted to stakeholders who have transportation, childcare, and flexible work hours. Education and outreach materials may only be presented in a single language, inhibiting stakeholders who speak and read different languages from fully participating in planning and governance. These are some of the reasons why communities continue to struggle with environmental justice issues associated with water resource management. The USEPA defines environmental justice as “the fair treatment and meaningful involvement of all people regardless of race, color, national origin, or income with respect to the development, implementation, and enforcement of environmental laws, regulations, and policies.” As evidenced by events including, but not limited to the Flint water crisis in Michigan in the US (Hanna-Attisha et al. 2016), the consequences of the inequitable distribution and management of water resources can be profound.

Participation by members of local communities also can be especially important in relatively undeveloped areas where resource conflicts affect the livelihoods of indigenous populations. In remote regions, scientific study may be very limited, making local knowledge all the more valuable. In areas where indigenous people or local communities of long-standing have strong cultural associations with and livelihood dependency on natural resources and the state of the environment, these individuals and communities are not only key stakeholders, but also valuable sources of knowledge. Traditional ecological knowledge (TEK), also called indigenous ecological knowledge, increasingly is seen as a complement to scientific knowledge, especially in areas where scientific data are scant. Although capturing all facets of TEK in a short definition is difficult, TEK is typically defined as a cumulative body of knowledge, practices, and beliefs, handed down through generations by cultural transmission, concerning the relationship of living beings (including humans) with one another and with their environment (Berkes et al. 2000; Failing et al. 2007). In some ways TEK is similar to the local knowledge that can be provided by a number of identifiable groups, including long-time community residents, indigenous people, and resource users with specialized knowledge such as fishers, farmers, or hunters. Usher (2000) suggests that the knowledge of indigenous people is likely to be richer, however, covering a larger region and accumulated over a longer time, and consequently the breadth of aboriginal environmental knowledge and the scope for drawing connections among phenomena may be greater.

Traditional ecological knowledge has been incorporated into environmental flow assessments in remote locations with a strong indigenous presence. The Patuca River, Honduras, Central America, flows through three national protected areas that include the roadless territory of two indigenous groups, the Miskito and the Tawahka, who depend on riverine and riparian ecosystems for navigation, agriculture, artisanal fisheries, bush meat, edible and useful plants, and drinking water. To develop environmental flow recommendations to mitigate effects of hydropower development in a data-poor region, Esselman and Opperman (2010) held workshops with representatives of the indigenous communities where participants annotated photographs and used hand-drawn maps to show water levels associated with different river conditions, how different flood levels affected crops and communities, and the most challenging passage points for boat traffic. The ability of indigenous fishermen to recognize taxa, behavioral traits, and spatiotemporal changes in fish assemblage composition across seasons was consistent with findings from the fish biology literature. Flow prescriptions for this data-poor region, based on hydrologic analysis, research published in the scientific literature, and local knowledge included low flows for each month, high-flow pulses, and floods, in dry, normal, and wet years (Fig. 15.12).

The threats posed to indigenous people's belief systems and livelihoods can lead to very strong opposition to a development project. In the Altai region of Siberia, a combination of TEK and cultural beliefs stopped a large dam and changed the course of development in a region. The Katun River in the Altai region of Siberia contains large numbers of important cultural sites dating from the Neolithic, and is considered central to the culture of the indigenous Altaians. Studies by cultural anthropologists describe spiritual beliefs related to the river, including curative sacred springs, special words said while crossing the river, and avoidance of taking water at night, which may upset the spirit of the river (Klubnikin et al. 2000). Proposed construction of a large dam would have significantly altered the ecology of the region, and the Altaian people would have lost much of their sacred and cultural landscape. Opposition to this project united indigenous people, well-known Siberian writers, and scientists in a protest that successfully defeated plans to build this 80-m high dam. Today, all of the major rivers of the Altai Republic remain free of dams, new buildings stress energy efficient construction, and energy is supplied by renewables including solar, wind, and small-scale hydro (https://www.altaiproject.org/).

Finally, it is important to note that the advances of recent decades have occurred mainly in rich countries. A global analysis comparing human water security threat with biodiversity threat found stark contrasts between rich and poor nations (Vörösmarty et al. 2010). Much of the developed world faces the challenge of protecting biodiversity while maintaining established water services. In contrast, the developing world faces threats to both human water security and biodiversity. In many parts of the world, environmental protection is less robust because resources are fewer and improvements in livelihoods are more urgent. The world's large rivers and their floodplains are home to some 2.7 billion people. For many of these river systems, human pressures are intense, and economic dependence on these river basins for water, power, and food is high (Best 2019). In such settings, river basin management will need to focus on co-benefits of simultaneously supporting economic development while protecting biodiversity and key ecosystem functions (Poff and Matthews 2013).

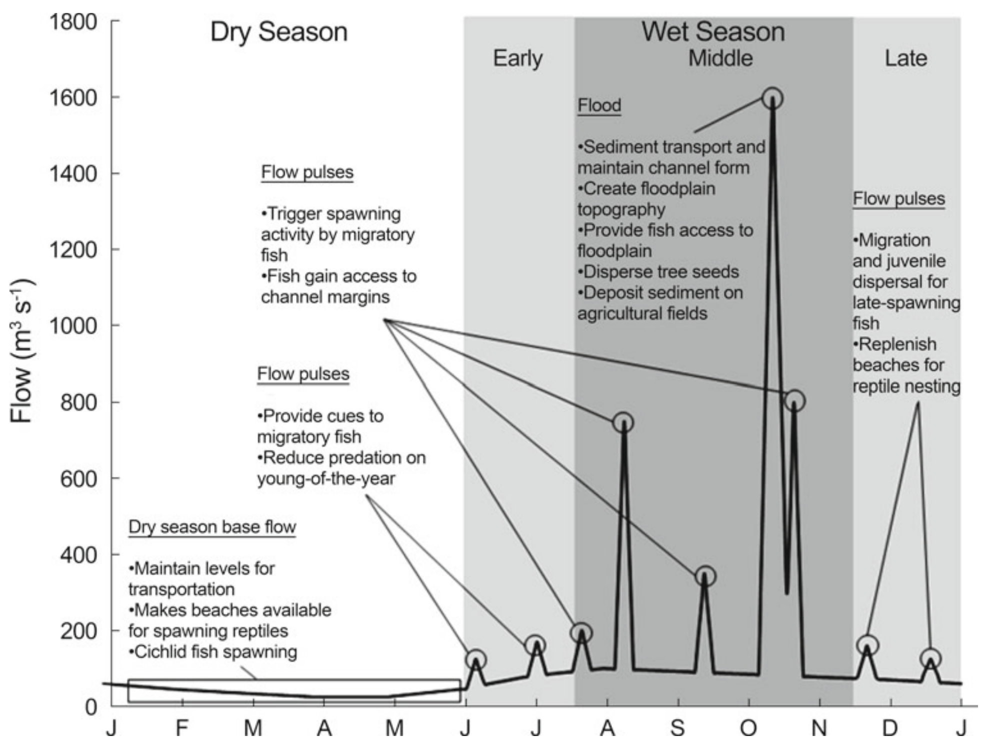

Fig. 15.12 A graphical summary of environmental flow recommendation (dark black line) for the Patuca River, Honduras, in response to a proposed hydropower dam. Dry and wet seasons are indicated by white and gray shaded areas, respectively. Environmental flow components for a "normal" hydrological year. (base flow, high flow pulses, and floods) are labeled with some details of the important ecological and social values that they support. Ideally, the timing of flow pulses and floods will be adjusted as a function of reservoir inflow, rather than having the static shape presented here as an example. (Reproduced from Esselman and Opperman 2010)

15.5 Progress Made, Progress Needed

The urgent need to repair, restore, and conserve river ecosystems around the globe is beyond question. Rivers face an abundance of threats, and many of these threats appear to be on an upward trajectory. Biological diversity is severely imperiled. Surveys of river condition find that most are in poor or fair condition, especially in developed countries. The success of restoration activities is poorly known and at best, mixed. Land-based protected area design may be inadequate for freshwater ecosystems. Change is desperately needed.

It is important, however, not to lose sight of what has been accomplished. As a thought exercise, consider the progress of the past two decades, using the year 2000 as an inflection point. Threats to rivers and the imperiled status of freshwater biodiversity were highlighted in papers by Allan and Flecker (1993), Malmqvist and Rundle (2002), and Strayer (2006), among many others. Ecological restoration became the primary objective of a great deal of stream management, resulting in dramatic growth of published research and projects described as restoration activities (Bernhardt et al. 2005). Emphasis has shifted from restoring structure to restoring function, and there is wide recognition that local actions will be most successful if embedded in coordinated actions at catchment or river basin scale. Increasingly, these are carried out within a formalized planning process (Fig. 15.7). At about the turn of the century, the United States and countries of the European Union underwent a major change from an emphasis on effluent water quality standards based on pollutants, to recognition that the condition of waterbodies should be assessed using biological and ecological measures. Although biological assessment has a long history, and some major methodological advances date from earlier decades, widespread adoption and standardization of biological assessment has taken place within the past two decades (Barbour et al. 1999). Scholarly recognition that fresh waters had received inadequate attention in conservation planning can be seen in papers published around the year 2000 (Saunders et al. 2002). At about this time, a focus on ecosystem services gained prominence and sparked advances in defining and quantifying human benefits (Millennium Ecosystem Assessment 2005). Public engagement and stakeholder participation have become more widely recognized as critical to project success. Finally, as described throughout this third edition of *Stream Ecology*, myriad research advances have greatly improved out understanding of the basic principles that govern the structure and function of stream and river ecosystems.

No doubt, for every advance just described over the past two or so decades, one can find antecedents in excellent work performed over many decades of the 20th century. But that should not obscure the main point of this thought exercise: repairing, restoring, and conserving river ecosystems is a long, complicated process, and not much time has elapsed since the urgency has been widely acknowledged. This is important to keep in mind when confronted with the legitimate and important argument that progress to date has been disappointing. Looking forward, we can hope to see continued progress in every area, but three seem especially important to call out.

First, better understanding of the ecological success of management interventions is urgently needed. This is widely recognized (Palmer et al. 2014; Wohl et al. 2015; Friberg et al. 2016), and it is sobering to realize how little information exists to assess the overall success of restoration projects, the European Union's WFD, USEPA's TMDL program, or freshwater conservation planning. One explanation is simply the time needed for system recovery. Ecosystems may recover slowly, and improvement may become more evident in future years. Another is the challenging task of assembling reports of widely varying detail from thousands of projects carried out by different agencies and states. These issues can be addressed through improvements in monitoring, reporting, and the establishment of common data banks, and likely would be especially helpful for lessons learned about actions at the local or reach scale. Although the science and practice of monitoring have undergone major advances in the past several decades, the need remains to develop more targeted methods able to detect gradual improvements and to improve diagnosis of the cause of deterioration (Carvalho et al. 2019). A greater challenge lies in the need to understand how local projects integrate at the catchment or river basin scale, and how these collective actions may be at the mercy of unaddressed threats within and beyond the catchment. The emphasis on integrated river basin planning within the WFD is a welcome signal. However, determining how best to design a suite of complementary projects at the catchment scale, and then assessing their collective success, is uncharted territory. Conservation planning approaches may help, but are intended to develop the optimum suite of targets, rather than the optimum suite of threat-ameliorating actions. Because so much of the real work of river management is carried out by agencies, there is a need for cross-jurisdictional collaboration, aided by institutional arrangements that ensure common goals, methods, and programmatic frameworks informed by best available science. Finally, resources are limited, and so it is important to be able to identify a suite of actions that is sufficient to achieve a desired level of river condition, justifying the return on investment.

Second, the influence of a changing climate will manifest in many ways that are poorly understood. Our ability to forecast future river flows is affected by what has been called the end of stationary, meaning our ability to predict future flows, floods, and droughts based on the historical record. Flow prescriptions intended to benefit the biota become more uncertain, possibly requiring a shift towards prescribing a range of flows that can sustain resilient, socially valued ecological characteristics in a changing world. Climate change will influence temperature regimes as well as overall runoff and flow regimes, with consequences for extreme events, seasonal timing, and all of the metrics describing flow magnitude, frequency, duration, and so on. In turn, the composition of riparian vegetation, rates of productivity, organic matter decay, and biogeochemical cycling will be altered. The full complement of changes can at best be speculated upon. Climate change also is expected to impact societies by changes to water runoff and supply, increasing water stress in some areas and hazard risk from flooding in others (Palmer et al. 2008). These impacts are likely to be greatest in river systems already affected by dams, relative to free-flowing rivers, and require a range of management interventions that may or may not benefit the ecosystem. One can hope that the global community will act to slow climate change. Managers of river ecosystems must prepare to adapt to its consequences through a deeper understanding of its impacts and appropriate actions that enhance system resiliency.

Third, greater appreciation is needed that what is being managed is not just an ecosystem, but a socioecological system. Humans are an integral part of virtually every ecosystem, directly and indirectly contributing to their degradation, benefiting from and enjoying their services, and serving as the source of the human capital to study and advocate for their protection. Promising steps have been made in public participation, stakeholder engagement, and incorporation of traditional ecological knowledge in planning and decision-making. A change in emphasis is underway, from a limited focus on meeting human needs while attempting to mitigate environmental costs, to an emphasis on sustained human benefits, including water for direct human use and water to support healthy ecosystems and the services they provide. It will be necessary to go one important step further. Protection of rivers is served by a shared societal vision of what a healthy river provides, aesthetically as well as functionally. Appreciation for the views and sounds of nature, and of rivers, is universal. To return to an earlier theme, people will protect what they love. The greatest challenge in protecting rivers is to vastly increase the numbers who care about protecting rivers. The future of river conservation lies in a full accommodation of human needs and values into a shared and equitable vision of the healthy rivers and streams that we wish to bequeath to future generations.

15.6 Summary

In the age of the Anthropocene, the world's rivers are amongst the planet's most highly altered ecosystems. Freshwater biological diversity is highly threatened, as much or more than terrestrial counterparts for taxa where comparative data exist. Yet, healthy streams and rivers benefit humans in myriad ways. They provide drinking water and harvestable fish, generate hydropower and supply irrigation water when harnessed by dams and canals, are useful for navigation, and absorb flood waters. In addition to these tangible benefits, running waters have aesthetic values that include the pleasures people experience from fishing, paddling, or strolling along a riverbank, but extend much further into the spiritual realm.

There are many reasons why society should repair, restore, and protect river systems for present and future generations. One important rationale centers on ecosystem services, the goods and services that an ecosystem provides free of charge, and the benefits that people receive from ecosystems. These can be grouped into provisioning services, such as fish, drinking water, and hydropower; regulating services such as waste decomposition and water purification; supporting services, including basal resources, nutrient cycling, and habitat provisioning; and cultural services, including educational, recreational, aesthetic, and spiritual benefits. An ecosystem services perspective makes explicit that benefits may be lost due to environmental stressors, and also provides a way to message and quantify the benefits gained under ecological restoration. Quantification of ecosystem services relies on a mix of methods ranging from direct market valuation to indirect methods assessing peoples' willingness-to-pay for a benefit and proxy estimates.

A second rationale, arguably more fundamental than service valuation, emphasizes the importance of nature to human well-being. This includes the spiritual and psychological, the restorative experience that derives from encounters with natural environments, and much that is difficult to put into words. This is the love of nature broadly, and of rivers specifically, which we call rheophilia. Support for this perspective can be found in scholarly analysis of human responses to landscapes and sounds, from measurements of mood, and in the art, literature, poetry, and music that draws inspiration from and celebrates natural settings. It should be noted that such thinking is not restricted to the most wild and remote images of nature, but also includes rural and urban landscapes in which people live their lives. The restorative power of nature is not limited to a natural world with no human presence, but is enhanced by experience with the sights and sounds found in more natural settings.

These two rationales explain the "why" of repairing, restoring, and protecting rivers, providing motivation and urgency. The "how" blends river science, human perceptions and beliefs, socio-economics, politics, and much more. Management actions aimed at improving rivers increasingly emphasize a holistic approach that attempts to create or maintain some aspect of river form and function that aligns with hydrologic, geomorphic, and ecological processes. This can be accomplished by relieving pressures that degrade and harm a river system, thereby promoting natural recovery; and by active measures to assist recovery that may include dam removal, addition of habitat elements, control of an invasive species, ensuring environmentally beneficial flows, and many more such actions. In recent decades, such actions increasingly are referred to using the term river restoration, and can be characterized by a more explicit pairing of science and practice, and by goals focused more strongly on recovering historic form and function to the extent feasible. There has been lively debate over the appropriateness of benchmarking against historical, undisturbed condition, whether form or function is the more suitable perspective, and the success or lack thereof to date.

Preceding chapters in this book describe the fundamental science that provides the understanding and the toolkit for managing rivers. Here we emphasize the frameworks for implementing management actions, under three headings. Integrated river basin management (IRBM) recognizes that the river basin (catchment, watershed) is the appropriate scale for organizing water management, because water sources and uses in a watershed are interrelated. Because political boundaries rarely correspond with watersheds, watershed-scale decision making requires institutional arrangements that provide for cross-jurisdictional cooperation. Enactment of the Water Framework Directive with the goal to attain good ecological status for waters of the 28 member states of the European Union, in force since 2000, has led to re-invigoration of IRBM and significant improvements in science, environmental monitoring, and formalization of the planning process in river management. In the United States, the Clean Water Act requires states, territories, and tribes to set water quality standards, which, if not met, require development of a Total Minimum Daily Load (TMDL) to address the cause of failure to meet designated uses. Rather distinct from these agency-driven approaches to remedy degraded waters, the field of conservation planning attempts to identify rivers systems, portions of river systems, or locations important for their habitat elements and species, to be designated as protected areas. Often driven by conservation organizations and supported by governments and international conventions, these efforts seek to ensure representation of the most biologically important representatives of ecosystems and habitats. While freshwater conservation planning has lagged behind similar efforts in the terrestrial realm, this field is advancing rapidly, developing frameworks that take into account the longitudinal and lateral connectivity that characterize rivers, as well as their vulnerability to external threats.

Successful management actions to repair, restore, and protect rivers will require expertise from many sectors, confidence that actions taken are likely to produce desired results, and support from the public and institutions. These are the three pillars of river management: fundamental science, measurement of progress, and societal support. There is realistic fear that river ecosystems, so highly threatened by human impacts, will continue to decline. There also is justifiable concern that efforts to date have failed to deliver the hoped-for gains. But the acceleration of knowledge, concern, and effort is relatively recent. Ecosystems are complex entities with many interacting parts, such that system responses to management action generally contain some element of uncertainty. There is still much to learn about the timeline and pathway followed by recovering ecosystems. Continued efforts are called for.

Further maturation of river management should build on lessons learned, through improvements in monitoring, reporting, and the establishment of common data banks. More work is needed to understand how local projects integrate at the catchment or river basin scale, and how these collective actions may be at the mercy of unaddressed threats within and beyond the catchment. Lastly and most importantly, greater appreciation is needed that what is being managed is not just an ecosystem, but a socioecological system. Humans are an integral part of virtually every ecosystem, directly and indirectly contributing to their degradation, benefiting from and enjoying their services, and providing the human capital to study these ecosystems and advocate for their protection. A change in emphasis is underway, from a limited focus on meeting human needs while attempting to mitigate environmental costs, to an emphasis on sustained human benefits, including water for direct human use and water to support healthy ecosystems and the services they provide. Protection of rivers is best served by a shared societal vision of what a healthy river provides, aesthetically as well as functionally.

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