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Prospects for sustaining freshwater biodiversity in the 21st century: linking ecosystem structure and function

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Biodiversity in freshwater ecosystems is under grave threat from human activities, due to the combined effects of multiple stressors such as pollution and habitat degradation, flow regulation, overfishing, and alien species. Consequently, a higher proportion of freshwater species are threatened to extinction than their terrestrial or marine counterparts. While this indicates the degree to which current practices are unsustainable, the actual situation is even worse as a failure to take account of shifting baselines has led to underestimation of historic declines. Anthropocene trajectories of rising human population growth and water consumption will be exacerbated by climate change impacts and consequential environmental alterations which, in combination with existing stressors, will lead to further extinctions. Such losses seem likely to impair ecosystem functioning and hence provision of goods and services that underpin human livelihoods. Unfortunately, evidence of a close relationship between biodiversity and ecosystem functioning (B–EF) is insufficient or equivocal at present, and B–EF science is not sufficiently mature to allow detailed predictions of precise outcomes of biodiversity loss or management needs for fresh waters. In the face of such uncertainty, it would be prudent to adopt the precautionary principle and minimize further losses. Despite the need for additional B–EF research and more effective communication of the importance and value of freshwater biodiversity, it is imperative that scientists and stakeholders collaborate to apply existing, albeit incomplete, knowledge to mitigating impacts and implementing conservation, management and restoration strategies in an adaptive fashion.

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Introduction

The Convention on Biodiversity (CBD) is the first attempt to obtain a global-scale commitment to conservation and sustainable use of biodiversity. Upon its implementation in 2002, participating nations agreed to achieve a significant reduction in rates of loss of biodiversity by 2010. Hopes of meeting this target — which coincides with the UN International Year of Biodiversity — were dashed upon publication of the Millennium Ecosystem Assessment (MEA) in 2005, which stated baldly that use of most of the ecosystem services upon which humanity relies was unsustainable, leading to '... substantial and largely irreversible loss of the diversity of life on Earth' [1].

Recent global analyses of progress towards the 2010 targets show that indicators of the state of biodiversity (e.g. extinction rates, population trends) are trending downward, with no signs of any reduction in rates of decline, whereas indicators of threat intensity (pollution, over-exploitation, invasive species) are increasing [2,3]. There is also disturbing evidence that humans have overstepped planetary boundaries for essential Earth-system processes [4,5]; crossing these biophysical thresholds could have detrimental or even disastrous consequences for humanity and ecosystems. Of seven boundaries quantified, three appear to have been transgressed: those relating to climate change; alteration of global nitrogen and phosphorus cycles; and rates of biodiversity loss [4,5]. A 'relatively safe' boundary of biodiversity loss (i.e. one comparable with historical rates of extinction) is being exceeded by at least one to two orders of magnitude [4,6]. Accelerated loss will be associated with alterations in ecosystem functioning that impair provision of goods and services and limit potential adaptation to further change [3,7,8]. Of additional, great concern is that withdrawals of surface runoff (blue water) have reached a level ($\sim$4000 $km^3\,yr^{-1}$) where the consequences of further withdrawals could include collapse of freshwater ecosystems as well as major shifts in the global water cycle [4,5]. Consumptive use of blue water is less than withdrawals ($\sim$2600 $km^3\,yr^{-1}$), but could increase by up to 50% over the next 20 years because of the need to enhance food security for growing human populations [4,5].

This selective review describes declines in freshwater biodiversity, the threat factors and vulnerabilities that have contributed to such declines, and the possible implications for freshwater ecosystem functioning and human livelihoods. Strategies that may contribute to limiting declines and mitigating impacts on ecosystems are also set out. Particular attention has been paid reviewing current literature (2009–2010) on the link between freshwater biodiversity and ecosystem functioning, and the influence of environmental change on that relationship.

Freshwater biodiversity at risk: the big picture

With respect to the global water cycle [9] — and some, perhaps many, other Earth system processes — we are now in the Anthropocene epoch (*sensu* Steffen *et al.* [10•]) where humans are the dominant driver of environmental change. Global demand for water has increased four-fold over the last 50 years [1] and the anthropogenic footprint on fresh waters is evident from the complete collapse of some ecosystems (e.g. the Aral 'Sea'), the gross pollution of others (e.g. the Ganges River), and the well-known instances of rivers that fail to flow to the sea every year (e.g. the Colorado, Murray-Darling, Yellow, and Indus). Unsurprisingly, the impacts on freshwater biodiversity have been severe [11–13] and far exceed whatever margins would have been sustainable for freshwater biodiversity — a trend acknowledged in the MEA [14] but one that has yet to impinge upon public perception to an extent comparable to awareness of threats facing the marine or terrestrial realms [3,6,15,16]. Furthermore, since estimates of planetary boundaries for rates of biodiversity loss are derived from data on marine and terrestrial species only [4], they underestimate freshwater losses. Indeed, since 1970, populations of freshwater animals have declined more than twice as fast as their counterparts on land [11,17]. There is growing realization that, in terms of extinction risk, freshwater species are among the world's most threatened biodiversity [12,13,18]. In addition, there remains an unquantified, but probably substantial, extinction debt due to anthropogenic activities with consequences that have yet to play out completely [13] and declines that occurred in the past which have been overlooked. Although not considered herein, loss of genetic diversity (e.g. by the extirpation of locally adapted populations) is likely to be widespread in fresh waters, and thus the extent of total biodiversity losses (species plus genes) will have been underestimated.

Shifting baselines

Since medieval times in Western Europe, fish and other freshwater animals (especially beaver and unionid mussels) have experienced historical declines beginning around 1000 AD [19,20]. Weir and mill construction further impinged on populations of riverine or migratory species and, combined with agriculture and land-use change, exacerbated the consequences of overexploitation leading to reductions in mean size and abundance [19,20]. Allowing for some differences in scale or intensity of action, these are much the same factors that threaten freshwater biodiversity today. From 17th and 18th centuries, migration of Europeans led to overexploitation of fauna in parts of the world that had hitherto been influenced only by indigenous peoples [21•]. All of these impacts occurred well before any formal stock assessments, giving rise to the false impression that conditions in the immediate past reflect conditions in the intermediate and distant past: that is deception due to a 'shifting baseline' [21•]. Losses of migratory fishes (alewives, eels, salmon and shad), unionid mussels and keystone species such as beaver from North American rivers are certain to have influenced nutrient transfer, food webs and organic-matter dynamics [21•,22–25]. Aside from the tendency to underestimate the extent of human impacts, the shifting baseline misleads those involved in restoration efforts, resulting in a failure to take account of historical losses that had functional significance for ecosystems and hence require replacement [21•,25].

Current threat status of freshwater biodiversity

The reasons why inland waters are hotspots of endangerment are straightforward. Firstly, the disproportionate biological richness of freshwater ecosystems makes them extremely vulnerable to species loss. The ~125 000 freshwater species that have been described represent 9.5% of known animal species on the planet, including one-third (>18 000 species) of all vertebrates, even though inland waters cover <1% of the Earth's surface [12,26••]. Secondly, a high level of threat impinges on freshwater biodiversity due to the interaction of an array of human-mediated process or stressors: pollution; flow modification and over-extraction of water; destruction or degradation of habitat; over-exploitation of fishes and herpetofauna; and impacts of alien species [11,12,18]. Thirdly, the degree of vulnerability and the intensity of threat are magnified by the fact that fresh waters are embedded in a terrestrial landscape where they serve as 'receivers' of wastes, sediments and pollutants in runoff, yet their limited volume lacks the capacity to dilute contaminants or mitigate other impacts [12].

The latest revision of the International Union for the Conservation of Nature (IUCN) *Red List of Threatened Species* [22] provides a reliable indication of the current threat status of freshwater biodiversity. It classifies 37% of all freshwater fishes and 31% of amphibians as threatened, with comprehensive assessments showing that European and North American fishes are particularly at risk [28,29]. The intensity of threat is especially striking given that freshwater fishes comprise a quarter of all vertebrate species, and virtually all amphibians breed and spend part of their lives in freshwater. Knowledge of the conservation status of freshwater invertebrates is inadequate [11,12,30], but there is little scope for optimism: for instance, 48% of North American crayfish species are estimated to be threatened [31]. IUCN assessments have been made for only two invertebrate groups globally: dragonflies and freshwater crabs [32,33]. While only 14% of dragonflies are threatened, the proportion rises to 40% if species classified as 'data deficient' (DD: essentially, those that are too rare to assess population trends adequately) are considered to be at risk [27]; the proportion of threatened freshwater crabs increases from 16 to 65% with the addition of the DD category. Nor is this insufficiency of knowledge confined to invertebrates: 25% of amphibians are classified by the IUCN as DD. Combining this figure with the percentage known to be threatened reveals that 56% of amphibians are either imperiled or too rare to assess!

Data deficiency and inadequate knowledge of freshwater biodiversity in tropical latitudes [11,12,26••] is a sign that the overall extent of threat could be even greater than described above, and is particularly disturbing given that in 2006, 10 000–20 000 freshwater species were estimated to be extinct or imperiled [30]. Among notable examples of decline are freshwater fishery stocks, especially large river fishes and particularly those species that undertake breeding migrations, most Asian freshwater turtles, and suites of endemic molluscs in North America [12,13,25,30,34,35]. The recent declared extinction of the Yangtze River dolphin, *Lipotes vexillifer* [36] and the failure of extensive surveys to detect any Yangtze paddlefish, *Psephurus gladius* [37•] — the world's longest freshwater fish — are particularly significant. They demonstrate that the shifting baseline is not only a matter of historic interest, as surveys along the Yangtze show that rapid cultural baseline shift has changed the perception of these large and charismatic species within a human generation. As soon as the Yangtze dolphin and paddlefish failed to be encountered on a fairly regular basis, they were rapidly forgotten by local communities and even by some individuals occupied in fishing [38]. This breakdown in expectations of what species should be present in fresh waters, and hence societal loss of interest in their preservation or conservation, has been dubbed 'ecosocial anomie' [25].

Impacts of multiple stressors on freshwater biodiversity

Impacts on freshwater biodiversity are among the consequences arising from what the scientific committee of the Global Water System Project (GWSP) has described as a 'pandemic array' of transformations in the global water cycle, including changes in physical characteristics, and biogeochemical and biological process in freshwater systems which, together with rapid shifts in human water use and withdrawal, are causing dramatic changes in patterns of water stress [39]. In the Anthropocene world, almost all types and sizes of lotic and lentic freshwater ecosystems are subject to multiple stressors [40–43] at a range of temporal scales [42,44] within complex spatial settings [45] including riparian zones and terrestrial surroundings [41,43]. To make matters worse, transgression of planetary boundaries for climate-change and nutrient cycles [4,5] reduces water quantity and quality for humans and biodiversity through the effects of elevated temperatures on the global water cycle and the water needs of agriculture, as well as eutrophication arising from elevated nitrogen and phosphorus loadings [45]. Feedback between stressors such as climate change and nutrient loading may produce unexpected outcomes and temperature increases beyond any predicted thus far [46].

Interaction among stressors can be exemplified by the tendency of disturbed or degraded ecosystems to become susceptible to invasion by alien species, which can alter the hydrology, biogeochemical cycles and biotic composition of invaded ecosystems thereby modulating the impacts of other stressors [47•]. Degraded or artificial habitats, such as reservoirs and man-made lakes, can support multiple invaders and act as a stepping stone for the spread of aliens to natural lakes [48]. Increases in forest area for carbon sequestration to offset climate change could also increase soil water use and reduce river runoff that, in turn, will prompt construction of dams and reservoirs that will further alter natural flow cycles and degrade rivers [49]. This means that the ongoing global epidemic of dam construction and fragmentation of rivers by impoundments [50] not only has direct effects on biodiversity and material transfer but also facilitates aliens and their impacts on native species. Greater frequency of extreme events such as floods and droughts under global climate change will likewise encourage water-engineering responses and exacerbate impacts on freshwater biodiversity [13,49].

Linking species loss to ecosystem functioning

Arguments for the protection of biodiversity tend to hinge on the assumption that all species contribute to healthy ecosystem functioning and hence sustain provision of goods and services for humans. Widely practiced attempts to assess the environmental health of fresh waters are, in fact, predicated on the notion that there is a close relationship between habitat conditions or water quality and the diversity of aquatic assemblages. For instance, biomonitoring programs in a number of temperate countries make use of fishes, diatoms or (most commonly) benthic macroinvertebrates to compare the extent of alterations in assemblage structure and species loss with reference conditions and thereby obtain an indication of the extent of habitat degradation or impairment [51,52]. Both explicitly and implicitly, then, the link between assemblage structure (i.e. biodiversity) and environmental health (i.e. ecosystem functioning) is assumed to be fundamental to much of the activity undertaken by freshwater scientists and water managers, even though the evidence for the link between species loss and functional impairment has yet to be well established.

Attempts to estimate a planetary boundary for biodiversity loss are constrained by the likelihood that species are not equally important for ecosystem functioning which, in turn, limits the usefulness of setting a 'safe' extinction rate. To what extent does species loss from freshwaters affect ecosystem functioning and their ability to provide ecosystem goods and services for people? As an example of what is at stake, at least two billion people depend upon rivers directly for the provision of ecosystem services that can be characterised most simply as 'food' [53]. The livelihoods of around 500 million of these people have been blighted because dams 'flatten' the seasonally fluctuating river flows that enhance biodiversity and determine established patterns of fisheries, flood-recession agriculture, and dry-season grazing.

Evidence that freshwater ecosystem functioning can be impacted by changes in biodiversity began to accumulate during the last decade [54,55]. The nature and magnitude of impacts depend, to a great extent, on the properties of the biodiversity–ecosystem functioning (B–EF) relationship and is therefore a matter of fundamental concern for predicting the outcomes of species loss [7,8,56]. Unfortunately, understanding of the B–EF relationship across scales is incomplete at best. A group of experts recently compiled a list of 100 questions which, if answered, would have a major influence on the policy and practice of biodiversity conservation [57]. Eight questions concerned ecosystem function and services, and six of these addressed the B–EF relationship. In addition, three of five questions devoted to freshwater ecosystems addressed aspects of the B–EF link [57].

A summary of the competing possibilities for the B–EF relationship must include the 'conventional' view which states that ecosystem functioning is enhanced or stabilized in a near-linear fashion as species richness increases, and *vice versa* (the diversity-stability hypothesis). A second possibility is that loss of species has no effect on function until some critical threshold below which the remaining species can no longer compensate for the loss of others and complete failure may occur (the redundancy or rivet hypothesis). A third possibility is that the B–EF relationship is unpredictable: functioning may be unaffected by the loss of certain species, but greatly impacted by the loss of others (e.g. keystone species such as salmon or beaver [22–24]). This idiosyncratic hypothesis holds that the identity of species lost is crucial (i.e. composition is critical, and keystone species may be present), whereas the number remaining is of secondary importance. Which, if any, of these descriptions applies to B–EF relationships in freshwaters?

Recent research in freshwater biodiversity–ecosystem functioning

The answer is neither obvious nor likely to be simple. The variety of possible effects is reflected in organic-matter processing in streams, especially the dynamics of leaf-litter breakdown mediated by microbes (primarily fungi) and shredding detritivorous invertebrates, and the effects of leaf and detritivore diversity on breakdown rates [58••,59,60•]. Increasing detritivore richness most commonly accelerates breakdown, although neutral and antagonistic effects are also seen at similar ranges of richness variation, indicating that multiple mechanisms underlie the B–EF relationships [58••]. Effects of litter mixing can likewise be positive or negative (or lacking) depending, in part, upon the extent to which the attributes of component species in mixtures vary [61], or may switch from positive to antagonistic effects as interactions proceeds [62]. In addition, changes from facilitation effects to apparent redundancy can occur as species richness of fungi in mixtures increases [58••]. Overall, the species composition or identity of species in mixtures — whether they are leaves, detritivores, or microbes — seems to have a greater influence on litter breakdown than species richness *per se*, thereby supporting the idiosyncratic B–EF hypothesis [58••,59,60•]. Simulations of species loss indicate that functioning responds more strongly to the sequence of loss rather than the number of species remaining [58••].

Greater realism in design of B–EF studies of organic-matter processing is needed with respect to selection of species mixtures (so as to incorporate species most vulnerable to anthropogenic change); enhanced trophic and/or functional complexity in experiments; and investigation of biodiversity effects on other ecosystem functions beyond litter breakdown [58••]. Transition of laboratory tests of B–EF relationships to field settings and increasing the duration or time scales of experiments also remain as major research challenges, as does expansion of the scope of investigation of B–EF relationships in freshwaters to include a greater array of functional measures (i.e. other than leaf-litter breakdown). To date, exploration of other B–EF relationships in fresh waters suggests the same pattern of compositional and idiosyncratic relationships observed in B–EF studies of litter breakdown, with a prevalence of redundancy in some instances [8,63,64]. Because much about the B–EF relationship remains uncertain, it would be prudent to adopt the precautionary principle and minimize further losses, thereby maintaining the option value of biodiversity and ensuring the persistence of poorly known species that may be critical to goods and services [65].

Human impacts on biodiversity and freshwater ecosystem functioning

A parallel line of research that has accompanied B–EF studies addresses the question of whether structure or function provides a 'better' measure of the effects of human impacts on the integrity of freshwater ecosystems. It is by no means obvious whether stressors will affect structure or function, or both [66]. For instance, reductions in biodiversity will affect function if the species lost are redundant or do not contribute to the function of interest [56]. The application of litter breakdown as a proxy for functioning has originated within the last decade [67] and, although other proxies are possible, such as primary or secondary production, nutrient cycling and whole-stream metabolism [68,69], it has since been applied widely [70–73] probably because it is relatively simple to measure.

Suggestions that the equivocal performance of functional proxies may not justify the added expense of their use over and above structural measures [71] can be countered by the evidence of changes in functioning along gradients of anthropogenic impact [67,72–74]. There are also instances where impairment is detected by changes in function with little or no shift in structure [73,74], and cases where function posts an early warning of structural change [79•]. Such findings mandate inclusion of functional measures in assessments of ecosystem health and integrity [56], and further research on the relationship between structural and functional measures will provide insight into the likely consequences of biodiversity loss on ecosystem function.

Much remains to be learned, but the results of B–EF research and structure versus function relationships have the potential to improve the understanding of how anthropogenic stressors modulate ecosystem functioning and thereby affect the supply of goods and services for humans [8]. In most instances, however, we have insufficient understanding to predict how species loss will affect ecosystem functioning, and which species are most vulnerable to individual stressors or their combination. That said, it is apparent that large (and especially migratory) fishes are highly vulnerable to human impacts [12,25,34,35] with the consequence that they and other large aquatic vertebrates become so scarce and their ecological roles are degraded to an extent that they might as well be extinct: that is they are 'functionally extinct' [36]. Larger species targeted by fishers tended to have disproportionate effects on nutrient cycling, and extinction simulations from Rio Las Marías (Venezuela) and Lake Tanganyika show that nutrient recycling is dominated by relatively few fish species so that the ecosystem consequences of loss, which differ according to whether nitrogen or phosphorus is of interest, depend on species identity [64]. This lends credence to the view that B–EF relationships are dominated by idiosyncratic or compositional effects, especially in ecosystems where keystone species are present [22–24]. Moreover, the likelihood that loss of large species has strong effects on ecosystem functioning gains support from evidence of alteration of carbon dynamics by overfishing and dams in Rio Las Marías that depleted a dominant detritivorous prochilodontid [75]. The loss of this species from lower trophic levels indicates a lack of functional redundancy in carbon dynamics but, in terms of nutrient cycling, a compensatory response in remaining species moderated extinction impacts [64], signifying the existence of functional redundancy among fishes. Thus redundancy or lack of it, and hence properties of the B–EF relationship, vary among functional processes — even within a single river.

Sustaining freshwater ecosystems with environmental flows

Most of our understanding on the B–EF relationship has been gained at small scales, with the few exceptions noted above. There is thus a critical disjunct between the scales at which the patterns and processes are understood and the scale at which management occurs and policy is set [76], and it is not certain how or to what extent small-scale processes operate at larger scales. Environmental flow (e-flow) allocations provide one instance where we have some confidence that the gap between scales of ecological understanding and management can be bridged. The e-flow concept, as defined by the GWSP [39], is 'the quality and quantity of water necessary to protect aquatic ecosystems and their dependent species and processes . . . in order to ensure sustainable development of water resources'. The success of conservation and restoration efforts will depend upon understanding and accurately modeling relationships between hydrological patterns, fluvial disturbance and ecological responses in rivers and floodplains [77,78] and, implementing water allocations that fall within a range set by the inherent resilience of these ecosystems [78,79•]. Determining the most appropriate method (among the very many available) for achieving this aim has not been straightforward.

A considerable amount of recent research has been devoted to 'an invigorated global research programme to construct and calibrate hydro-ecological models and to quantify the ecological goods and services provided by rivers in contrasting hydro-climatic settings across the globe' [80], thereby helping to achieve the water-related goals of the MEA as well as, incidentally, addressing the e-flow challenges identified by the GWSP. A degree of consensus among e-flow practitioners has emerged recently around the Ecological Limits of Hydrologic Alteration (ELOHA) approach for determining regionally relevant hydro-ecological models and water allocations [81••]. Although details of e-flow methodologies lie beyond the scope of this selective review, an important new development to note is the incorporation of Bayesian hierarchical modeling to increase the inferential strength of assessments of e-flow effectiveness. Such models overcome the shortcomings of weak statistical design arising from a lack of replicate sample units (rivers) in most e-flow implementations, and maximise the benefits derived from large-scale e-flow monitoring [82] which is especially beneficial considering the paucity of studies with adequate monitoring before and after e-flow implementation [76]. Bayesian decision networks have also been used to compare the relative benefits of restoration options such as e-flows in the all-to-common situation (see Impacts of multiple stressors on freshwater biodiversity) where freshwaters are influenced by multiple stressors [83].

In addition to ELOHA-type determination of e-flow allocations *a priori*, improved *a posteriori* management of dams to mitigate their downstream impacts can achieve partial restoration of natural river flow and temperature regimes so offering a potential application for e-flows in ecological restoration [84–86]. Other initiatives include development of 'holistic' methods that strike a balance between development and resource protection to determine water allocations among multiple users; these have been applied in parts of Africa and Asia where a paucity of data on hydro-ecological relationships rules out the ELOHA approach [87,88].

Whichever approach is adopted, implementation of e-flows must be combined with strategies for engagement with water managers, local communities and other stakeholders through a process of consultation and adaptation [89], and water allocations must be conducted as rigorous, large-scale experiments within an adaptive management framework involving scrupulous analysis of outcomes [53,89,90]. This evaluation is essential: meta-analysis has shown that restoration of physical habitat (the 'if you build it, they will come' approach), although widely used in stream restoration, does not result in ecological recovery if the stressors limiting restoration have not been alleviated [91].

Prognosis and prescription: where to from here?

Although e-flows offer some hope of enhanced sustainability and improvements in well-being for people and ecosystems, the extent of endangerment of freshwater biodiversity offers a reliable indicator of the degree to which current practices are unsustainable [12]. It remains a major and urgent challenge to find satisfactory ways to manage inland waters for multiple uses, and to restore or rehabilitate already damaged systems taking due account of the need to adjust restoration targets in the context of the shifting baseline [25]. Climate change intensifies that urgency. Because the risks from business-as-usual and management failure are so large, freshwater ecosystems should rank foremost among the environmental priorities for science funding [18]. In that context, a new European Union initiative to build a global information platform that will host databases on the distribution, trends and status of freshwater biodiversity on a global scale (BioFresh: www.freshwaterbiodivesity.eu) is most welcome. However, even in the ideal situation where more resources were available for research, we do not have the luxury of time. Research, management and conservation must thus proceed in tandem, and be combined with public engagement which, among other things, will be essential for continued funding. How can this be done?

First, we must do a better job of communicating the importance and value of freshwater biodiversity, since scientists have failed to convey the significance biodiversity losses to humans. (After all, if we had done so, these losses would not have continued at such a rate.) The emerging Intergovernmental Science Policy Platform on Biodiversity and Ecosystem Services [92] could provide an opportunity to convey that message. At the same time, we must continue to research the B–EF relationship at least until we attain some predictive ability based upon it. Second, we must ensure that our research findings are actually used (or are, at least, potentially useable) in real-world settings, so that scientific knowledge can be linked with action [13,93]. Ensuring that this takes place will be exceptionally challenging. Communication alone is unlikely to be an effective means to influence drivers such as the growing need for water, hydropower and so on. It will need to be combined with engagement, environmental advocacy, and empowerment of stakeholders with adequate information.

The Healthy Waterways Programme in South East Queensland, Australia [94•], is a paradigmatic example of what is achievable within a freshwater monitoring initiative, which reports on ecosystem health at a regional scale and guides investments in catchment protection and rehabilitation. In this environment, multiple stressors, acting at different spatial and temporal scales, interact to affect water quality, biodiversity and ecosystem processes. Sixteen variables are measured twice a year at 250 sites, and are summarised in an annual report card that is presented in a public ceremony to local politicians and the broader community. Identification of the primary causes of degradation and the appropriate spatial scale for rehabilitation or protection allow targeted investments in catchment management, greater public confidence that limited funds are being well spent, and better outcomes for freshwater ecosystem health [94•].

The Healthy Waterways Programme has effectively leveraged the scientific approach to environmental management by adding a social and political dimension to a research and monitoring programme in order to build consensus over what can and should be done to protect fresh waters. A similar partnership approach in the north-eastern United States, involving a diversity of stakeholders, has resulted in novel land-use and management practices to achieve the joint aims of economic development and protection of vernal pools that serve as amphibian breeding sites [95]. These two examples suggest that application of technical tools and knowledge to areas that have traditionally been seen as beyond the ambit of researchers will likely be key ingredients of the success of future endeavours to protect biodiversity and ecosystems [6]. An additional ingredient is a willingness to advocate for policy outcomes that arise from research done. This may create new problems of ensuring objectivity and independence of researchers [95], and care will need to be taken to ensure that such advocacy does not compromise the reputation of scientists as trusted and objective sources of expert knowledge [96]. However, this expertise does not diminish scientists' obligations as citizens or stewards of nature, and a number of 'best practice' approaches to effective science-based advocacy that can inform policy recommendations have been identified [96,97].

What next? Research needs to be salient to the concerns of stakeholders, the information provided must be that which is needed (since much scientific information is never applied), and the source of the information must be credible. A realignment is needed in which researchers and stakeholders collaborate to reach decisions about the multiple uses of water since this will increase the contribution of science to problem solving and sustainable management. The prognosis for freshwater biodiversity will thus depend on our success at establishing collaborations that ensure the best use of existing — albeit insufficient — scientific knowledge.

Conflicts of interest

The author declares that there are no conflicts of interest.

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