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Linking ecology with social development for tropical aquatic conservation

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ABSTRACT

1. Tropical aquatic ecosystems are species rich, with high numbers of endemics. Increasing pressure from human activities, including demands for food and energy, growing human population, and economic aspirations, highlights the need for a more concerted effort towards aquatic conservation.

2. Understanding of tropical aquatic ecosystems has developed largely from a northern temperate perspective that might not be always appropriate. Applying classic models of how water bodies function can hinder effective conservation strategies. This is coupled with very incomplete knowledge of species distributions and their ecology.

3. Better understanding of tropical aquatic ecology to guide conservation needs a research agenda that connects more strongly with the social-ecological realities of tropical ecosystems.

4. Although approaches to conservation may be contested, a fundamental challenge to protection of aquatic habitats is a lack of capacity at the individual and institutional level. Without this, the development of improved techniques and approaches for tropical aquatic conservation will fail to reverse current trends of degradation. Research outputs on tropical aquatic ecosystems remain dominated by institutions based outside the tropics.

5. Building awareness and practice to conserve the aquatic ecosystems of the tropics can be supported through extending the dialogue across sectors and by connecting tiers of governance. An ecosystem services framework that identifies the benefits that humans derive from ecosystems provides a powerful tool, often linked with estimates of economic value. However, this can neglect important regulating services or distract from more fundamental existence value.

6. The preservation of tropical aquatic diversity will only be achieved if recognized as important at all levels, from local to global. Targeted external support can build awareness and capacity, but conserving aquatic ecosystems requires local commitment. Developing community monitoring that provides straightforward information on ecosystem health presents opportunities to connect citizens with the ecosystems that, ultimately, they depend on. Copyright © 2016 John Wiley & Sons, Ltd.

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INTRODUCTION

Limnology and oceanography, and later links with conservation science, were built on the foundations of European, and North American, academia and culture. Early documenting of tropical biodiversity relied on spatially extensive expeditions (Worthington, 1996), usually involving a type of hit-and-run approach. It is not surprising that perceptions of ecosystem function and the philosophy of aquatic conservation in the tropics mirrored early models developed in the US and Europe, but not necessarily suited to the character of the tropics. How conservation is, or should be, guided in tropical countries is an increasingly important question. Tropical aquatic ecosystems contain both high species diversity and number of endemic species (Gaston, 2000), yet often lack financing for environmental research, monitoring, and policy-making processes, and local societies often possess low levels of formal education. The challenge of implementing effective conservation is further compromised by limited knowledge of ecological functioning and species distribution of tropical aquatic ecosystems (Darwall *et al.*, 2011).

In this article, we attempt to identify key issues of importance for tropical aquatic conservation. This highlights: (a) the increasing pressures on tropical aquatic ecosystems; (b) how the nature of tropical aquatic ecosystems can be informed by, and differ from, ecological models and management approaches developed in temperate zones; (c) the policy paradigms that shape the conservation of those ecosystems; and (d) the principal needs for improving aquatic conservation in the tropics.

Pressures and policies

Freshwater habitats are documented as both the most globally diverse per unit of area and the most threatened and degraded (Dudgeon and Smith, 2006; Darwall *et al.*, 2011). International targets on halting biodiversity decline have been largely unsuccessful (Millennium Ecosystem Assessment, 2005; CBD, 2014). The rate of ecological change affecting aquatic ecosystems in tropical developing countries is likely to be much greater than in temperate zones (Darwall *et al.*, 2011; Holland *et al.*, 2012). The extent of this, however, is often poorly documented. A critical assessment of species and habitat distributions, and the relationship with aquatic conservation in tropical regions are pressing needs.

Throughout the tropics there is a large range of multiple and interacting threats to biotic integrity (Dudgeon *et al.*, 2010; Van Asselen *et al.*, 2013) that includes needs for enhanced food production, accelerating urban development, over-exploitation of inland and coastal fisheries, pollution from industry, land-use changes, species introductions, and disruption of aquatic connectivity from, for example, dam construction. These adversely affect tropical biodiversity and ecosystems in multiple and interacting ways (Vörösmarty *et al.*, 2010).

The need to conserve tropical biodiversity, and the importance of aquatic ecosystems to underpin human well-being, is well established (World Commission on Environment and Development, 1987). That urgency was reaffirmed by the UN Sustainable Development Goals (SDGs). Target 6.6 of the SDG on Water states 'By 2020, [to] protect and restore water-related ecosystems, including mountains, forests, wetlands, rivers, aquifers and lakes'. Recently, the US National Intelligence Council (US National Intelligence Council, 2012) concluded that sustainable water management presents the largest global security threat. The Asian Development Bank (ADB, 2013) highlights water challenges with the catchphrases 'Too Much Water, Too Little Water' and 'Too Dirty Water'. African economic development policies for water explicitly recognize the importance of environmental quality (European Commission, 2011) and in South America inter-governmental initiatives support water conservation, although only Colombia has policies specifically targeted towards aquatic ecosystems (Castello and Macedo, 2016). Meanwhile, in the headwaters of the Amazon, controversial concessions for exploration of oil and gas remain sanctioned by the governments (for example) of Bolivia and Peru (Finer *et al.*, 2008). Globally agreed policies for the protection of the environment and its biodiversity often appear disconnected from *de facto* practice. The reasons for this undoubtedly reflect a complex mix of societal actions and governance at local to medium scales, and a lack of ecological knowledge at the appropriate political scale. Nevertheless, if the decline of habitat quality and loss of species is to be reduced, a more concerted effort is needed to link quality of ecosystems with sustainable development and mitigate rampant pressures.

An informative example of these problems is the Ramsar Convention on Wetlands, which came into force in 1975 (www.ramsar.org) and covers a large range of water bodies, including coral reefs. The Convention is an intergovernmental treaty, committing signatories (that now stand at 169) to the conservation of wetlands. While it may have reduced the rate of decline of wetlands, and facilitated other conservation efforts (Holland *et al*., 2012), degradation and loss of wetland area has continued even where there are apparently strong national policies for wetland protection and in specifically designated Ramsar sites (Finlayson, 2012; Davidson, 2014). In many countries designation as a Ramsar site may, however, be the only designation that offers any protection for wetlands, but many sites lack effective management or monitoring.

In many developed countries, severe environmental pressures are frequently controlled through legislation backed up with effective implementation, and supported with industry-funded monitoring through licensing. Even in developed countries, challenges such as addressing diffuse pollution, invasive species, and better integration of environmental and economic policies are prevalent. In the tropics, the challenges posed by under-resourced or reduced human capacity of scientific and managerial institutions are exacerbated by increasing population densities, the need for economic growth, and increasing impacts of more variable and unpredictable weather patterns. Global trade and food security add further pressures on both inland and coastal waters.

Although approaches to conservation of tropical aquatic systems can be informed by experiences from elsewhere, it can never be that simple spatial transference can be expected to be successful. Ecological processes, and resilience and resistance of tropical biotic communities to stress, may not mimic temperate counterparts. Therefore, there is a need to link ecological and social understanding and practices in order to inform conservation practice and the often difficult ethical debates that go to the philosophical and practical heart of tropical conservation (Dowie, 2009; Sarkar and Montoya, 2011). A key question that remains unanswered is what are the essential features that need to be better understood to achieve successful tropical aquatic conservation?

THE NATURE OF TROPICAL AQUATIC ECOSYSTEMS, STABILITY AND CONSERVATION

With seasonality shaped by patterns of precipitation, more consistent temperatures, and often unique geological history, many characteristics of tropical ecosystems are intrinsically different from temperate ones. Most obvious is a general trend of higher species diversity and greater specialization and adaptive radiation among many aquatic groups. This raises speculation on whether this might also affect ecosystem functioning and stability. High species diversity and its relationship with trophic structure, coupled with limited detailed research, makes ecologically informed conservation of many tropical water bodies extremely difficult, and a solid base of ecological knowledge is often limited. Unlike Alice in her wonderland (Carroll, 1865), running faster to stay still is not fast enough, so developing and disseminating general principles to support the knowledge base for tropical aquatic conservation is of primary importance.

Rivers and streams

Any overview of tropical rivers and streams clearly has to recognize the extremely large variation in ecological character and the difficulty in making comparative generalizations among, especially, the larger tropical rivers and very different biogeographical regions. The annual cycle of flooding of the Amazon, for example, discharges 20% of the world's fresh water into the oceans, and in its 6400 km journey from the headwaters to the delta encompasses waters with very different chemistry and productivity. For many large rivers of the tropics such as the Mekong and the Ganges, large-scale degradation of upland ecosystems and altered river hydrology have affected many ecological processes. Conservation efforts still remain predominantly focused on damage limitation rather than, increasingly needed, restoration (Dudgeon, 2005). Rivers such as the Congo have suffered much less impact, related to many decades of civil and political unrest, although at local scales unregulated activities such as mining have caused serious degradation. Nevertheless, and particularly for smaller rivers, is it possible to draw out some general principles important for ecosystem function and aquatic conservation?

The almost ubiquitous paradigm of stream and river ecosystem functioning is the River Continuum Concept (RCC) (Vannote *et al*., 1980), developed from observations of northern temperate rivers. The model is predicated on the concept that under natural conditions an equilibrium of energy transfer is established, fashioned through longitudinal succession of invertebrate functional feeding groups. The headwaters, dominated by allochthonous inputs processed by a predominance of shredders, give way to greater prevalence of collectors and then grazers as the river moves from heterotrophy to autotrophy in the more open waters downstream. While there have been varying views as to the generality of the RCC, the model has been used as a basic descriptor of tropical streams and rivers. In a previous special edition of *Aquatic Conservation: Marine and Freshwater Ecosystems*., (2006, vol. 16, issue 7) devoted to 'Conservation of Tropical Streams' Moulton and Wantzen (2006) put forward the proposition that at least as a first approximation the principles of ecosystem functioning of temperate rivers and streams could be applied to the tropics. As so many temperate rivers have been extensively modified by humans over the last few centuries, however, their use as models for the tropics can be questioned.

In northern Australia, for example, rivers in more than 80 drainage basins are mostly not dammed and free of impacts (Douglas *et al*., 2005). At least in that 'wet–dry' tropical climate, understanding their functioning can help develop approaches to tropical conservation elsewhere. The rivers are strongly influenced by the variable hydrology and lateral continuity with floodplains, and food webs linked to algal production rather than macrophyte and allochthonous carbon (Douglas *et al*., 2005; Jardine *et al*., 2012). Thorp and Delong (2002) developed a Riverine Productivity Model for tropical streams, highlighting the importance of algae in the food chain of otherwise heterotrophic rivers, and in recent years there has been a rethinking of the importance of allochthonous material in the food chain of small, shaded, tropical streams, with conflicting views on the role of shredder taxa (Camacho *et al*., 2009; Li and Dudgeon, 2009). A few macro-consumer species often dominate the benthic food webs, and omnivory and short food-webs are common.

A descriptive approach based on mouthparts, feeding behaviour and gut contents may also misrepresent animals' trophic status and ecosystem function. Using temperate keys to assign invertebrate functional feeding groups may have underestimated the role of shredders in tropical rivers (Dobson *et al*., 2002; Cheshire *et al*., 2005), and stable isotope analysis suggests that apparent shredders and detritivores are trophically linked to microalgae and not to the detrital material of their gut contents (Douglas *et al*., 2005; Lau *et al*., 2009). Such organisms may be functional shredders, acting as 'ecosystem engineers' in the sense of causing a large bioturbation in the system, but can also be dependent on algal-consuming prey. In addition, it is clear that large grazing mammals such as hippopotami can have profound effects on river nutrient dynamics and its connection with the floodplain (Pennisi, 2014; Subalusky *et al*., 2015).

The importance of flood pulses for lateral connectivity with floodplains, ecosystem dynamics and conservation is well recognized (Junk, 1999; Mosepele *et al*., 2009) and a feature that sets tropical systems apart from temperate ones, where the pulse is frequently less dramatic and largely controlled through catchment drainage and engineering. Many tropical floodplains are crucial for ecological processes and survival of species that rely on seasonal inundation or migrations enabled by such connectivity (Moulton and Wantzen, 2006). Limited understanding, and hence definition, of floodplain dynamics can, however, limit inclusion into both conservation and land-use policies (Ellery and McCarthy, 1994; Junk *et al*., 2014). The importance for conservation is further emphasized because of the role of many tropical floodplains in supporting human livelihoods. Continuing and often rapid changes to lateral ecosystem connectivity, including changes in land use, and decimation of mammal herbivores or their replacement with domestic cattle, can seriously affect energy budgets and ecosystem functioning of tropical streams (Masese *et al*., 2014).

Impact on river connectivity is most notable through severance by dams or pollution. These have both local and regional consequences not only for migratory species but all taxa because of changes in ecosystem functionality related, for example, to access to floodplains and spatial changes in hydrology (Pringle, 1997; Greathouse *et al*., 2006). This can be disastrous both for conservation and human livelihoods (Crook *et al*., 2015).

Wetlands: swamps and marshes

The term ‘wetland’ has been applied to a broad variety of aquatic systems, so that a coherent model of ‘wetland ecosystem functioning’ is impossible. Here we focus on those wetlands with a permanent or semi-permanent swamp or marsh-like character, often with distinct hydrogeomorphological units (Maltby, 2009). They cover a range of habitats from upland peatlands to shallow lakes surrounded by fringing emergent vegetation typified by plants such as *Phragmites* spp. or *Papyrus* spp.

Tropical wetlands encompass huge areas, such as the Pantanal and Amazonian floodplain in South America, wetlands of the four major African river ecosystems (Nile, Niger, Zaire, Zambezi), the 119 Ramsar sites in India, Tram Chim, and many other areas of Indochina, Alligator River and Gulf of Carpentaria in Australia. Extensive as some sites are, they nevertheless represent small vignettes of a larger but diminishing resource (Davidson, 2014), many of which remain undocumented or ill-defined (Finlayson, 2012; Junk *et al*., 2014). In Brazil, the diversity of wetland types has hindered wetland inventories (Junk *et al*., 2014). If classification of wetland types is not achieved, effective monitoring or management is unlikely to follow. Wetlands provide a range of ecosystem services from highly productive provisioning of food and vegetation to important regulating services including sequestration of carbon and water retention reducing flood risks (Maltby and Acreman, 2011). Even small wetlands provide a spatially interconnected network of habitats, and associated, and multiple, human benefits across a nested range of scales (Mitsch and Gosselink, 2000; McCulloch *et al*., 2003).

The hydrology and associated water chemistry of wetlands lie along physical and chemically defined continua and include shallow permanent lakes, to palustrine, riverine, lacustrine or floodplain marshy areas, and ephemeral and endorheic water bodies. Tropical wetlands are often major breeding and roosting sites for resident and migratory birds, often because of their exceptional productivity (McCulloch *et al*., 2003). While the understanding of temperate and sub-tropical wetlands (Maltby, 2009; Mitsch and Gosselink, 2015) provides a framework for the classification and description of tropical analogues, the hydrology, ecology, and social-ecology of tropical palustrine, riverine, lacustrine or floodplain wetlands is generally poorly understood. Many tropical wetlands are subject to very high amplitude seasonal inundation and extensive dry periods used for grazing and crop production (Verhoeven and Setter, 2010). Their scattered nature, often fringing lake and river systems, makes them particularly vulnerable to conversion to other uses, and potential human conflicts (McCartney *et al*., 2011).

Conversion of wetlands to agriculture is often seen as an attractive option owing to the combination of available water and carbon. This occurs at both small local and larger scales. The effect on sustainable food supply and nutrient enrichment of wetlands and downstream waters is, however, largely untested. Lack of field data makes it very uncertain what the net effect of conversions can have on greenhouse gas emissions (Pellerin *et al*., 2004; Masaka *et al*., 2014).

Observations of nutrient enrichment of tropical inland and coastal waters is of increasing concern (Yasin *et al*., 2010; Deegan *et al*., 2012), affecting other sectors such as off-shore fisheries (Ma *et al*., 2014). Although many tropical soils have highly depleted nutrient reserves, with many farming systems operating at negative nutrient balance (Schoumans *et al*., 2015), trends of intensification in wetlands alter vegetation structure and biodiversity. Increasing food supply in many tropical areas is a major international goal, but pressures to convert tropical wetlands could be alleviated with more focus on improving water retention and soil nutrient status in the drylands of a catchment (Falkenmark *et al*., 2007; Verhoeven and Setter, 2010). Increasing food by improving dryland crop yields (Critchley and Gowring, 2012) may be a more complicated approach than conversion of wetlands, but the wiser longer-term choice for both conservation and human well-being. Ambitions for increasing global rice production might also provide opportunities for conservation orientated cultivation (Pernollet *et al*., 2015), learning from traditional approaches (Linares, 1981; Lansing, 1987). Conversions of tropical wetlands, either for rice or other crops, also affects traditional use of tropical wetlands as common pool resources supporting complex arrays of local livelihoods (Verhoeven and Setter, 2010).

In many part of the tropics, natural wetlands are used for mopping up nutrient or heavy metal emissions. This represents a technology transfer in the use of constructed wetlands for effluent treatment, well developed in many northern countries. However, the use of natural wetlands to treat wastewater and farm effluent inevitably has limits, eventually leading to degradation of wetland ecosystems and increases in net nutrient export. The Nakivubu wetland fringing Lake Victoria provides a case in point. Used as part of a plan for mopping up pollutants in wastewater, large areas of the wetlands have subsequently been converted to agriculture, losing not only the potential for attenuation of nutrients but accentuating public health hazards when wastewater also contains heavy metals, pharmaceuticals and human hormones.

While many principles based on the knowledge generated in temperate zones can be applied to conservation of tropical wetlands, there are also extensive knowledge gaps, for example in the understanding of biogeochemistry and nutrient dynamics, sediment mobility from large river systems and the stability and dynamics of swamps. Of particular concern has been the global loss of mangrove forests (Polidoro *et al*., 2010). Degradation of mangroves through harvesting and development has direct consequences for coastal biodiversity and the human communities that depend on them either for livelihoods (e.g. from fisheries), or storm protection (Das and Vincent, 2009). The ecosystem functions of inland and coastal tropical wetlands, their ecosystem services, and the dynamics of the human social systems that depend on them require considerable further understanding (Giosan *et al*., 2014). Development of wetlands can be a false economy, with far-reaching effects on biodiversity and human communities connected socially to wetlands, or living in the protective shadow of (especially) coastal wetlands (Das and Vincent, 2009; Polidoro *et al*., 2010; Brander *et al*., 2013; Russi *et al*., 2013).

Lakes

The ecological interplay within a lake is choreographed within a stage of shape, size and climate. Depth interacts with temperature, salinity and prevailing wind to determine cycles of stratification and biological production. The interaction of climatic effects of temperature, light and wind on the structure and ecological functioning of lakes is, as in temperate lakes, fundamentally important. Understanding how physical structure affects nutrient dynamics, biotic structure, conservation and management also informs important differences between the northern temperate and tropical paradigms of lake ecology. Tropical lakes were generally not subject to the recent influences of glaciation that has so dominated the understanding of temperate ones, with their recent past of about 10–15 000 years. Lakes in the tropics are often much older and many of the deeper ones have been in existence for millions of years (Cohen *et al*., 1993).

In northern temperate climes, shallow lakes played a formative role in the early definition and concepts of ecosystem function (Lindeman, 1942), and later in that of their biological stability (Scheffer *et al*., 1993). This has guided approaches to conservation and management, and especially the biotic control of algal populations (McQueen *et al*., 1986). The alternation of clear water and turbid water so cherished by temperate water ecologists may not be an effective model for patterns observed in the tropics (Jeppesen *et al*., 2007). The well-studied and much debated patterns of plankton succession and its synchronization with fish recruitment are generally not relevant in tropical lakes. Trophic structure of secondary producers in tropical lakes tend to have smaller body size, with less impact on algae through size-dependent filtration rates, reducing the temporal importance of zooplankton grazing on the control of algae. The rapid turnover and high production rates, especially at the lower trophic levels, drive high production:biomass (P:B) ratios (Irvine *et al*., 2001).

The high and persistent seasonal temperatures that drive metabolic processes and rate of nutrient recycling, photosynthesis and algae turnover rates means that models developed for nutrient management in temperate zones need refining for the tropics. The impact of nutrient enrichment can occur at lower net nutrient loads, with the ecological consequences more severe (see Figure 9 in Lewis (2000)). This also applies to cycles of oxygen, whose saturation point diminishes with temperature. The link between increasing nutrient loads and greater and more persistent hypolimnetic oxygen deficiency is well demonstrated (Townsend, 1999), with particular consequences for deeper dwelling fish. Owing to higher overall seasonal temperatures, tropical lakes tend to stratify more readily and for longer periods than temperate ones. A greater likelihood for relative wind-driven heat loss causes frequent compressing and expansion of mixing in the upper layers and hence more seasonal variation in the depth of the epilimnion, allowing nutrient replenishment from the hypolimnion (Lewis, 2000). While the general paradigm applies (although with many exceptions) that temperate lakes are primarily limited by phosphorus, nitrogen limitation may be more important in tropical lakes. Dominance of blue-green algae is common, even at low nutrient concentrations (Lewis, 2000; Irvine *et al*., 2001), although their importance as fixers of nitrogen may overall be quite modest, with nitrogen fixation mediated by electrical storms of seasonal importance but with denitrification of potential major importance across a whole spectrum of water bodies.

In deep tropical lakes such as those of the African rift valley, water of the hypolimnion is permanently devoid of oxygen, yet during the cool season may only have a few degrees temperature difference from the surface water 200 m or more above. Recent evidence of small but consistent temperature increases in Lake Tanganyika have been proposed as a mechanism of nutrient depletion affecting fisheries production in the epilimnion because of a strengthening of the thermocline and reduced nutrient replenishment from upwelling (O'Reilly *et al*., 2003; Tierney *et al*., 2010). At the same time, evidence of reduced fish catch from intensification of fishing effort (Sarvala *et al*., 2006) highlights the need to consider interacting pressures in understanding these systems.

High, and often endemic, biodiversity adds an extra dimension for understanding ecological functioning and conservation of tropical lakes compared with temperate ones. The biogeography and long periods of geological isolation in many tropical lakes have enabled diverse species flocks to evolve. Although the cichlids, both in Africa and S. America, may have the highest general profile of endemic freshwater aquatic life, adaptive radiation leading to endemic species flocks has also occurred in other groups. In Lake Tanganyika, for example, these include assemblages of crabs, atyid shrimps, molluscs and ostracods (Coulter, 1991). Diversity of endemic, mainly cichlid, fish in Lake Malawi and Lake Tanganyika, and previously Lake Victoria, is particularly striking. The flexible body plan of the Cichlidae (Fryer and Iles, 1972) enabled dramatic adaptive radiation, resulting in species 'super-flocks' (Sturmbauer *et al*., 2010). In these lakes sympatric speciation driven by sexual selection, rather than as a consequence of (allopatric) isolation is a distinct possibility (Genner and Turner, 2005). Dramatic adaptive radiation of fish also occurred in Lake Victoria, although a large proportion of the 500 native cichlid haplochromines have been lost following the introduction of the Nile perch (*Lates niloticus*) in the 1950s. This has coincided with other environmental pressures, notably nutrient enrichment, but also pesticides and heavy metals, proliferation of the introduced water hyacinth (*Eichhornia crassipes*), and loss of fringing papyrus swamp (Verschuren *et al*., 2002; Sitoki *et al*., 2010).

While impacts on Lake Victoria may be the most dramatic for the African Great Lakes, evidence of nutrient enrichment, climate induced shifts in production, and decline of fish catches from over-fishing are generally widespread (Hecky *et al*., 2006). The largely endemic cichlid fish communities of the African Great Lakes are particularly vulnerable because of their low rates of fecundity, specialized diets and, often, restricted distributions associated with rocky outcrops. Across the tropics, fishing pressure is a major threat both to biodiversity and sustainable livelihoods (Allan *et al*., 2005). Commercial fisheries records show widespread increases in fishing pressure, and declines in catch per unit effort (CPUE) over the past five decades (Cowx, 2007). Widespread and increasing pressure for artisanal and subsistence fisheries may constitute as much pressure as commercial fisheries but, because fishing costs are often minimal and poverty widespread, fishing is not economically constrained (Castello *et al*., 2015). In Africa, inshore small-scale fishing such as beach seining degrades not only stocks, but habitat of sand-nesting cichlids. It has also driven reduction in mesh size and use of other illegal methods, including poisons (Crean *et al*., 2007).

In diverse multispecies fisheries where restricted distributions are common and recruitment of fish modest, extinctions can occur, but response of individual species and community structure vary with overall fishing pressure and gear selectivity (Welcomme, 1999; Allan *et al*., 2005). Increased fishing effort generally affects apex predators initially because of their large size, with cascading impacts on trophic dynamics, and nutrient recycling (McIntyre *et al*., 2007). Reductions in mean body size can increase extinction risk (Reynolds *et al*., 2001) and shift community structures as smaller species replace larger ones. In temperate zones, increases in fishing effort lead to increasing yields up to a maximum (maximum sustainable yields (MSY)), followed by subsequent decline with increasing effort. In contrast, in tropical multispecies fisheries increasing effort has been shown to lead to more constant yield with increased fishing effort beyond the MSY (Lorenzen *et al*., 2006), but associated with often dramatic reductions in the mean total length of harvested species (Welcomme, 1999; Castello *et al*., 2013). These types of changes suggest wider-reaching effects on the overall ecosystem. Reductions of molluscivore predators by fishing in Lake Malawi increased the prevalence of schistosomiasis among local human populations (Stauffer *et al*., 2006).

Throughout the tropics, the proliferation witnessed over the last four decades of reservoirs for water storage and energy generation is set to increase even further (Zarfl *et al*., 2015). This poses a number of major considerations for river flows and biotic migrations, affects downstream sediment and nutrient processes, and changes river morphology (Kunz *et al*., 2011). Within the new reservoirs, a shift from a riverine to lacustrine habitat affects ecological structures and functions. The creation of Lake Kariba in the Zambezi Basin in the 1950s provided suitable habitat for the introduction of the Lake Tanganyika sardine *Limnothrisssa miodon*, which developed into a highly productive fishery (Magadza, 2006). Escapes from Lake Kariba later found their way downstream into the Cahora Bassa reservoir, and a productive fishery established there. The management of tropical reservoirs also fundamentally affects water stratification and temperature regimes, with consequences for biogeochemical cycles and trophic dynamics as for tropical lakes (Lewis, 2000). High rates of mineralization, and tendency to accumulate organic matter can promote high methane releases, adding to greenhouse gas emissions (Yang and Flower, 2012), although the generality of this needs further work. The establishment of standing water in previous river systems can also lead to proliferation of water-borne human disease

(Jobin, 1999; Ziegler *et al.*, 2013) and invasive plant and fish species.

Suggestions in the 1970s to introduce *Limnothrissa* spp into Lake Malawi to fill a 'vacant niche' occupied by the insect *Chaoborus edulis* (Degnbol, 1990) were shown by later detailed investigations to have posed a high risk to the lake food web and fisheries (Irvine *et al.*, 2001; Darwall *et al.*, 2010). At a smaller scale, internal translocations of cichlids within Lake Malawi as a consequence of the ornamental fish trade have been shown to disrupt genetic sorting and integrity of localized endemic populations associated with rocky outcrops (Genner and Turner, 2005). As the world enters uncharted territory with respect to climate shifts, preserving the rich endemic communities of tropical aquatic systems offers possibilities not only for better overall understanding of adaptation of tropical aquatic communities, but a pool of species that can be an ecological buffer to ecosystem disruption.

Coral reefs

Although at a global scale the ecological and conservation importance of coral reefs are well recognized, the ecosystem services they provide can still be much underestimated by local stakeholders (Aswani *et al.*, 2012). Many early studies on tropical coral reefs applied paradigms of temperate community ecology, in particular the role of disturbance, stochastic and non-equilibrium dynamics, to the understanding of their functioning (Karlson and Hurd, 1993). In recent years there has been a development of new ideas of coral reef functioning and management (Mumby and Steneck, 2008; Aswani *et al.*, 2015).

Coral reefs, highly productive and efficient in recycling nutrients, have complex structure and intricate biotic interactions. The state of the ecosystem appears to be locally stable but subject to change of state or phase (Cruz *et al.*, 2016). As with phase shifts postulated in shallow lakes (Scheffer *et al.*, 1993), a major preoccupation of conservationists and managers relates to particular difficulties owing to the implied hysteresis of the process in that a return to an original preferred state does not happen by merely reversing the trend that caused the initial change (Cruz *et al.*, 2016). Although reefs may be resistant to change within the boundaries of the original state, restoration following a phase shift is much more complex. However, reefs are exposed to many stressors, often acting synergistically.

About 90% of shallow coral reefs occur in the Indo-Pacific, mainly in developing countries. For many, societal relationships with the reefs have been lost or neglected (Aswani *et al.*, 2015), while in other areas, particularly Micronesia, there are many good examples of effective traditional or more recently designed management (Richmond *et al.*, 2007). Coral reefs are highly vulnerable to pressures: a 300-year-old coral can be killed in hours to weeks, but may not be replaced for centuries. In the face of limited resources, management priorities often identify protected areas, guided by the distribution of species with the highest risk of extinction. In recent years this has been supported with new approaches. Molecular biology, for example, can identify specific cause-and-effect relationships, with molecular biomarkers able to identify proteins and enzymes produced by stressed corals and linked to specific pollutants. The application of these techniques, however, is inevitably limited by available funds in the majority of tropical countries (Aswani *et al.*, 2015).

Important for the conservation of coral reefs are their ecological links to other coastal ecosystems such as mangroves, seagrasses and the open ocean. Links with terrestrial ecosystems, particularly around islands, calls for a clear need to integrate management and the study of land and aquatic conservation. Eight out of 10 reef biodiversity hotspots and 14 of 18 centres of endemism analysed by Roberts *et al.* (2002) were adjacent to terrestrial biodiversity hotspots. The importance of ecosystem connectivity for recovery of damaged reefs is well illustrated in the Caribbean where extensive loss of mangroves can reduce fish populations in reefs (Mumby *et al.*, 2004; Pollux *et al.*, 2007). Mangroves also reduce runoff of sediments that can interfere with coral recruitment and growth (Mumby and Steneck, 2008). Marine protected areas may lose effectiveness unless coupled with terrestrial ones, which requires planning and legislation (Richmond *et al.*, 2007;

Russell *et al*., 2009), and new governance structures that embrace management of catchments with that of reefs (Mumby and Steneck, 2008).

Maybe more than any other wetland identified by the Ramsar Convention, coral reefs epitomize a need for a globally concerted effort. As well as the local or regional impacts such as over-fishing, invasive species, and pollutants from land-based activities, global ocean acidification and a shifting climate are of extreme concern for the persistence of many reefs. These effects are well illustrated by the Australian Great Barrier Reef (GBR). Recently the Outlook Report of the GBR Marine Park Authority (GBRMPA, 2014) concluded that ‘climate change, poor water quality from land-based run off, impacts from coastal development and some remaining impacts of fishing are the major threats to the reef’s future health’. The report concluded that ‘substantial reductions of pressures were required to prevent projected declines and improve the reef’s capacity to recover from the effects of climate change’. The Australian and state of Queensland governments have promised investment for improving water quality and habitat restoration. It is not surprising that for the largest coral reef system in the world there are multiple interests, from the point-specific industrial interests in port facilities and dredging, through to the widely distributed rural producers to the tourists and direct users of the reef. The connectedness with potentially geographically distant sources of pressures will necessitate commensurate engagement with multiple stakeholders. Whether such an approach can better safeguard the GBR in the face of other severe pressures such as the invasion of the crown of thorns starfish (*Acanthaster planci*) and climate-induced bleaching (Frieler *et al*., 2012) remains to be seen.

In some reef systems, such as those off the coast of Brazil, high turbidity from siliciclastic sediments (Leão and Kikuchi, 2001) and abundant plankton (Kelmo and Attrill, 2013) suggest a different evolutionary history. These ‘muddy water corals’ have a greater dependence on heterotrophy (Anthony, 2000), which may explain their apparent resilience (Miranda *et al*., 2013). Brazilian coral reefs are typified by low species diversity and high endemism (Nunes *et al*., 2008). It has been suggested that the more stressor resistant genotypes from these areas could be used to replenish other more remote reefs (Aswani *et al*., 2012), although translocation of species across reefs would merit careful consideration of costs against benefits.

CONNECTING TROPICAL AQUATIC ECOSYSTEMS WITH POLICY AND PEOPLE’S NEEDS

The richness of diversity of tropical aquatic ecosystems, longer evolutionary time for their development compared with temperate zones, and increasing pressures provide an urgency for developing effective mechanisms for their safe-keeping. Fundamental to this challenge is connecting conservation policy with people’s needs. While aquatic ecosystems provide services far beyond the supply of fish and other wetlands’ products, it is clear that developing mechanisms to reconcile local interests with ecosystem management requires a much broader perspective to conservation than has traditionally been the case (Zimmerer, 2000). Addressing these challenges includes better alignment of policies and potential to link with effective catchment and landscape planning. It involves discussions of the merits and social equity surrounding protected areas, and the potential of a more embracing social-ecological approach for effective conservation.

A number of international agreements set targets for conservation. The strategic 2050 vision of the Convention on Biodiversity Diversity (CBD) and the so-called Aichi targets call for sustainable use of ecosystems and maintenance of ecosystem services (CBD, 2012). The Strategic Plan is to ‘take effective and urgent action to halt the loss of biodiversity in order to ensure that by 2020 ecosystems are resilient and continue to provide essential services, thereby securing the planet’s variety of life, and contributing to human well-being, and poverty eradication’. Twenty headline targets for 2015 or 2020 are guided by five strategic goals (https://www.cbd.int/sp/targets/). The targets include sustainable harvesting of aquatic species, controlling invasive species, reducing pressures on habitats, preventing extinction of threatened species and implementing national action plans for awareness and protection of biodiversity. This key message is picked up by the UN Sustainable Development Goals (SDGs), but it is difficult to envisage reaching the relevant Aichi or SDG targets by 2020. While many countries have developed policies designed for the protection of aquatic ecosystems, effective implementation, and the resources needed for that, are generally lacking. Standards for water quality are highly variable across countries, with many loosely adopted from elsewhere. Finally, in what may be termed 'failed states', the finesse of conservation policy or practice hardly features within the considerations of government or formal institutions. Ironically, however, political conflict often reduces pressures on aquatic habitats.

In many tropical countries, the environmental degradation of the last half-century has been justified with reference to the logic of the Environmental Kuznets Curve (EKC)(Grossman and Krueger, 1991), such that attention to environmental protection becomes sequential, not simultaneous, with industrial development (Azadi *et al*., 2011). Critical analysis has questioned the applicability of the EKC (Mills and Waite, 2009), and the 'too poor to be green' argument is increasingly considered fallacious (Rudi *et al*., 2012). Nevertheless, environmental protection and conservation are often subjugated to national or local ambitions of industrial and agricultural development, or simply hindered by a preponderance of local impacts.

More recently a large and increasing literature makes powerful economic and business arguments for conservation of aquatic biodiversity (Russi *et al*., 2013; Costanza *et al*., 2014). With increasing global attention on 'green' solutions, this provides opportunities for developing countries to apply nature-based solutions for better and economically viable water management (Green *et al*., 2015). A key challenge to integrating biodiversity and its ecosystem functioning into the development agenda is how to reconcile that with continuing developmental pressures (Lucas *et al*., 2013; Van Asselen *et al*., 2013). Across large tracts of the tropics, food and energy security are priorities, with poverty alleviation often assumed a consequential benefit. As tropical countries strive for enhanced energy and food supply and populations continue to grow, conflicts with aquatic conservation will increase. The remark of Falkenmark (2004) that 'there are two particular imperatives to pay much more attention to in view of the evident conflicts of water-dependent interests: on the one hand food security, on the other hand ecological security' remains poignant. If habitats and their species are to be protected, conservation interests must consciously and actively work within the context of, and influence, the development agenda. Food and energy solutions that involve large-scale infrastructure projects such as dams and irrigation schemes often bring high risks for aquatic conservation, with ample evidence of undue optimism of the benefits to costs (Ansar *et al*., 2014). The enthusiasm for dam building witnessed in the Mekong, with serious consequences for both nature conservation and human livelihoods, is being replicated in Africa (Zarfl *et al*., 2015) and the Amazon (Castello and Macedo, 2016).

Development interventions that reduce aquatic biodiversity can also be a consequence of bilateral trade deals. While these often aim to bring rural communities out of poverty, they also pose direct or indirect threats to biodiversity (Laurance *et al*., 2014). Achieving effective trade-offs between development objectives and aquatic ecosystem health is a major challenge, not least because the merits of protected areas are increasingly contested. Over the last four decades, there has been a large expansion of protected areas for conservation in the tropics (Chape *et al*., 2005; Naughton-Treves *et al*., 2005), although less for aquatic than terrestrial habitats (Lovejoy, 2006; Nel *et al*., 2009a). Aichi Biodiversity Target 11 has the ambition to conserve by 2020 at least 17% of important biodiverse terrestrial and inland waters, and 10% of coastal and marine areas. While the Aichi targets make specific connection with equitable management and ecosystem services, critics of the protected areas concept point out the dilemma of conservation action that conceptually, or in practice, separates nature from people (Naughton-Treves *et al*., 2005; Sarkar and Montoya, 2011). Questions on the legitimacy of the persistence of conservation principles set in place in the 19th and early 20th centuries, and of social justice (Adams and Hutton, 2007), can collide with those of conservation biology leading to what Agrawal and Ostrom (2006) described as a 'dialog of the deaf'. The Protected Areas paradigm for nature conservation remains the dominant model in the tropics (Chape *et al.*, 2005; Naughton-Treves *et al.*, 2005), but has frequently led to a separation of refugia for wildlife from the wider countryside, and the resources used by citizens (Dowie, 2009). It is also the case that protected areas do not necessarily safeguard, especially, riverine aquatic ecosystems, because of the influences from upstream (Abell *et al.*, 2007; Nel *et al.*, 2009b). In the Amazon, for example, the protection of 56% of the basin's area has a relatively low effect on the maintenance of freshwater ecosystems that play pivotal roles in Amazonian livelihoods (Castello and Macedo, 2016). Indeed for many protected areas there is a lack of understanding of the extent that they safeguard freshwater species (Hermoso *et al.*, 2016).

Appreciating the interconnection of aquatic ecosystems and the merits of a 'wider countryside' approach to conservation provides a necessary counterpoint to the designated site-based approach. On both aspects, the quality of evidence and thinking has led in recent decades to major changes in global understanding of conservation biology, and the importance of spatial and temporal scales for species distribution and ecology. Technological innovations and landscape models have greatly improved the potential to identify priority conservation areas and strategic and systematic planning for aquatic conservation (Linke *et al.*, 2007; Nel *et al.*, 2009a; Nel *et al.*, 2011). Increasingly, conservation practice relies on remote sensing and GIS mapping that can document spatial and temporal patterns of land use, estimate water balances and identify connectivity among sites. Questions about which areas need to be targeted to meet criteria of representativeness, and incorporating considerations of irreplaceability, condition and vulnerability are increasingly informed by spatial models (Linke *et al.*, 2007). Sophisticated algorithms are not necessarily matched by effective field planning and management (Knight *et al.*, 2006). Identifying where conservation is most needed cannot by itself make it happen.

Across large swathes of the tropics, information on species distributions and their habitat requirements, water quality and quantity, and landscape geo-spatial data is missing or highly fragmented. Datasets, often held in various repositories, can lack coordinated processing. A lack of data sharing or incompatibility of collection methods restricts effective conservation assessment and management. Although concerns about reliability, ownership, or even international security, can restrict data sharing, these can be resolved through sound and collectively developed data policies. The need for more extensive and targeted monitoring of aquatic ecosystems necessarily requires improved technical capacity for managing datasets supported by GIS based Spatial Data Infrastructure (SDI). A number of international initiatives are in progress to develop better catalogues of spatial information to support decision-making for aquatic conservation and water management (CP-IDEA, 2013; USGS, 2015). This wisely includes a steady move away from reliance on commercial data management products to a greater use, and hence skill development, of open source software.

While technical developments have enabled conservation to be better informed about what and where to invest effort (Nel *et al.*, 2009b; Abell *et al.*, 2011), connecting the science of conservation biology with the understanding and dynamics of human social structures remains an additional and crucial issue, embracing a much wider spatial and societal perspective (Nel *et al.*, 2007). Conservation of aquatic ecosystems sits within complex physical and social networks. While sustainable resource management may align with conservation objectives, it is not an inevitable outcome, and the tenet that the protection of natural areas alleviates poverty is not always confirmed by empirical evidence (Roe *et al.*, 2013). The increasing realization of the importance of not only reconciling, but capitalizing on different stakeholder and institutional perspectives has led to advocacy for the necessity of a social-ecological approach to natural resource management and conservation (Norgaard *et al.*, 2009). Linking this to an ecosystem services framework helps focus greater attention on the wide range of benefits that aquatic ecosystems provide for human well-being. It also helps with the awareness and communication of the unseen ecological processes that underpin ecosystem quality. Although national economies and local livelihoods ultimately depend on the services and benefits that tropical aquatic ecosystems provide, making the clear and simple connections, especially beyond the more visible provisioning service, is sufficiently elusive that it has not stemmed the steady decline of aquatic ecosystem quality and extent. At a workshop on ecosystem services given by one of the authors of this manuscript (KI), participants familiar with Lake Victoria considered its ecological state to be of acceptable quality because it still provides a capture fishery important for food security. From the northern European perspective, the lake is highly degraded, and over-fished.

Establishing a new paradigm for conservation of tropical ecosystems based on an ecosystem services framework is a fundamental challenge to existing policy structures and edicts. It is also not without its dangers, as the ecosystem services framework, and its economic cousin of economic valuation, risk diminishing the importance of existence value (McShane, 2007). The challenges to traditional conservation approaches, especially those within the tropics, need to better embrace social and political science, to work through what remains a complex and largely fragmented range of views. Within this myriad of wisdom, voices from the south are still often largely absent, or constrained by capacity limitations, governance structures and cultural or political hegemony. The social-ecological reality of rural communities facilitates community engagement, but current capacities to train for sustainable development are limited. Investing in bottom-up support for communities to enhance awareness and share, and co-produce, knowledge, remains relatively underexplored. Greater integration of conservation and development policy is badly needed.

Furthermore, although it is a compelling argument that local resources should be managed by local communities (Sarkar and Montoya, 2011), this requires strong governance and institutions. The transition from a government-decreed ‘top-down’ approach to conservation, or natural resource management, to a community-based participatory model can lead to other issues of social justice as resources are captured by new elites (Lane and Corbett, 2005), or poor design of new governance structures fail to meet community expectations (Kahmann *et al.*, 2015). Addressing these issues requires deep understanding of human motivations, social inclusivity and the design of regulatory regimes to redress social or environmental damage. When sanctions on misuse of natural resources do not exist or regulation is not effective, tendencies for corrupt practices can become manifest. The corruption, like the conservation debate, is a complex one with links with nature conservation not always clear and certainly not ubiquitous (Barrett *et al.*, 2006). What is clear is that the rules and norms operating within a governance framework are of fundamental importance.

Governance structures provide the setting for effective and sustainable resource management and practice. As well as more obvious attributes such as accountability and the incorporation of technical advances, effective governance allows for adaptive management and facilitates learning within organizations (Pahl-Wostl *et al.*, 2007). Conservation strategies benefit most when rooted in evidence-based policy and adaptive management (Adams and Sandbrook, 2013), but are of little consequence unless connected with an enabling governance environment. It is also naïve to assume that, even with high quality evidence, decisions are necessarily made by rational actors. Both sustainable development and conservation requires skills that can communicate across institutional structures. This applies everywhere, but can be more pronounced within governance frameworks with limited resources or inflexible structure.

CONCLUSIONS AND THE WAY FORWARD

Given the recognized importance of aquatic biodiversity in both fundamental and utility terms, developing realistic mechanisms for its sustainability is essential. While some may argue that a long-term geological time frame renders such a discussion irrelevant, this ignores a responsibility to immediate and successive human generations. Human pressures on tropical aquatic ecosystems have caused major impacts in the last 50 years. The next 50 will test the extent that degradation and alteration of those ecosystems, and set within the context of climate change, leads to both ecological and human impoverishment. Reversing the trends of degradation will depend on a number of key factors that need to improve the connections between socio-ecological structures and scientific understanding, economics and capacity development. We conclude this article by highlighting four essential topics that require further understanding and action for the conservation of the biodiversity and ecosystem functioning of tropical aquatic ecosystems. Within each topic there are research, governance and educational needs. All topics, like the ecosystems themselves, are interlinked.

Topic 1: Complexity of tropical ecosystems and developing the knowledge base

The diversity and functioning of tropical aquatic systems are, compared with temperate waters, under-studied. Many questions remain on how tropical and temperate aquatic ecosystems differ in their ecological structure and function, and how that affects response to pressures. The social component that connects human livelihoods with aquatic conservation remains a major, and under researched, challenge, requiring a broad socio-ecological perspective that is able to learn from experiences across habitat and social contexts. Improvement in documenting species autecology and distributions remains a basic requirement. Linking that to ecological functioning of aquatic biodiversity requires better understanding and prediction of how organisms interact with catchment hydrology and respond to specific pressures. These include biological pressures, and while the impact of invasive species on aquatic standing waters in the tropics is probably widespread (Dudgeon and Smith, 2006; Pyke, 2008; Hecky *et al.*, 2010; Tricarico *et al.*, 2016), a systematic review of the topic appears lacking.

A search in the Wiley online library on 13 July 2016 of the terms ‘fish’ plus ‘fish’ AND ‘tropical’ as keywords produced 12 214 and 105 returns, respectively. A search of these terms in the titles of papers returned 16765 and 16 hits, respectively. Searches that replaced ‘fish’ with ‘invertebrates’ as keywords returned 1685 and 15 hits, and for titles 662 and zero hits, respectively. Filtering further by this journal’s name returned nothing for ‘tropical’ AND ‘fish’ as keywords and only three returns when titles of papers were searched. Similarly low scores were found for the journal *Freshwater Biology*. An earlier (22 May 2016) more general search of content using ‘fish’, and ‘tropical fish’, which searches for occurrence of both words independently in a paper’s content, in the home pages of these journals plus *Limnology and Oceanography*, returned much higher numbers of hits (e.g. 1314 and 438 respective hits for *Aquatic Conservation*) but a scan of first author affiliations indicated a preponderance of temperate based institutions. Furthering the understanding of the diversity and ecological functioning of tropical water still depends largely on support and funding from temperate based sources. This resource-limited reality inevitably restricts the depth of research that can be achieved in many tropical countries. Globally available databases can help. Examples include FISHBASE (http://fishbase.org), a valuable resource for distribution and habitats of many fish, and the Global Biodiversity Information System (http://www.gbif.org). The United Nations Environmental Programme (UNEP) has recently revived the GEMS water initiative (http://www.unep.org/gemswater) to provide access to global water quality data. Despite these types of initiatives, for most tropical countries obtaining data to assist with conservation management is limited. While there is a clear need for better data acquisition to help meet the challenges for protecting tropical aquatic biodiversity, and understanding the links with terrestrial systems (Raghavan *et al.*, 2016), this also opens important discussions on funding and support from richer countries. This can be troubled waters. The perceived wisdom of the need for knowledge transfer from wealthier to poorer countries needs to be reconciled with views such as those of Escobar (1996), that 'northern conservationists have no privileged status in the South'.

Too often, knowledge and conservation efforts in the tropics founder on a deficiency of enabling conditions that exist in developed countries. Instead of looking to establish what might be unrealistic in the short term, tropical aquatic conservation could be more effective if it followed the Indo-Chinese saying: 'when you don't have what you want, you make do with what you have'. An increasing number of studies show how local or traditional ecological knowledge can fill knowledge gaps, and reveal unsuspected patterns and ecological processes (Berkes *et al.*, 2000; Huntington, 2000; Motsumi *et al.*, 2012). Linking traditional wisdom with a verifiable scientific and evidence-based approach seems entirely common sense.

Recommendations

There is an urgent need for better documenting and understanding of tropical aquatic biodiversity and ecosystem functioning. Linking that to the social-ecological context of ecosystems necessitates not only a multidisciplinary approach, but a concerted alliance between natural and social sciences, and with full regard to the diverse and often conflicting views of local stakeholders. This requires time and patience. Without better integration of land and water management, any ambitions for an integrated approach to water and conservation will be thwarted. How this is all funded is a major issue and calls for effective mechanisms for capacity and skills development (see below), and in meaningful partnerships between the poorer tropical and richer, largely northern temperate, institutions.

Topic 2: An ecosystem services framework

Recognizing the benefits of ecosystem services, beyond the obvious provisioning of food, shelter and water, provides the means to their preservation. Although technical improvement (e.g. in spatial models) to target conservation actions can help conservation planning, it is not, fundamentally, a shortage of technical skills that threatens the future of tropical aquatic ecosystems The more pressing need is for the full range of ecosystem services that these systems provide to be incorporated into local and national decision-making. This, of course, extends beyond discussion only of aquatic conservation.

The ecosystem services framework (Millennium Ecosystem Assessment, 2005; Russi *et al.*, 2013) firmly brought the importance of ecosystems for human well-being into the political arena, but has not necessarily seen a beneficial consequence in many countries. Using an ecosystem services framework in formal procedures can be a powerful component of policy and decisions (Peh and Lewis, 2012; Russi *et al.*, 2013). Linking services to benefits (Fisher *et al.*, 2009) has been conceptualized well by the US EPA across the spectrum from provisioning to existence value (US EPA: http://www.epa.gov/aed/lakesecoservices/ecosl.html). As most decisions affecting aquatic ecosystems occur at local scales, there is an urgency to translate not just the philosophy, or rhetoric, but the methods across all tiers of government. Strongly linked to this is the means for quantifying and applying ecologically and socially acceptable environmental flow regimes (commonly known as 'eflows') for rivers (Tharme and King, 1988; Poff *et al.*, 2010), and more recently wetlands (King *et al.*, 2009). The eflows approach attempts to balance the needs of water flow to support the ecological processes in a river with local stakeholder needs. As pressures in the tropics increasingly alter river flows, there is considerable discussion on the need for an eflows approach, which is beginning to be incorporated onto the agenda of donor-funded projects. Although the methodology is well developed, its application was considered by Le Quesne *et al.* (2010) to be largely 'still at the stage of policy and debate rather than implementation'. Cursory and rapid assessment for eflows also risks legitimacy. The justification that any eflow is better than no eflow not only lacks conviction, but makes for poor science and policy. However, recent developments in Mexico on allocation of water regimes (flows) for wetlands is supported by legislation, under the 'Mexican law for the determination of environmental flow' and 'National Water Reserve Programme' (Programa Nacional de Reservas de Agua, PNRA), hence providing a regulatory approach to the maintenance of ecosystem services. In South Africa, linking wetland water allocation and quality assessment to wetlands has been undergoing similar developments (Kotze *et al*., 2008).

In general, greater attention on how best to exploit ecosystem services to avoid loss of natural capital is an urgent need for the coming decades (Costanza *et al*., 2014; Palmer *et al*., 2015). While it is difficult to attach monetary value to ecosystem services and there are practical and philosophical risks in doing so (Spash, 2011), it is nevertheless an increasing component of the ecosystems and conservation debate. Attempting an estimate of the annual global value of ecosystem services as $125 trillion (Costanza *et al*., 2014), must, by the very nature of the underlying assumption involve considerable uncertainty. The technique for estimating GDP in 2014 as little more than US$ 78 trillion for the globe (retrieved from www.wikipedia.org, 13 July, 2016), is also controversial. The key point is that economics can highlight the relative value of aquatic ecosystems to counter misguided assumptions that natural capital is a free service.

Irrespective of the difficulties in economic valuation of ecosystems, protecting functionally intact habitats generally costs significantly less than restoring degraded ones (Chen *et al*., 2009). Balancing conservation needs with human welfare is, in any case, never straightforward and identifying effective trade-offs is complex (Arthur *et al*., 2004; McShane *et al*., 2011). A review by Blignaut *et al*. (2013) reported that only 3% of restoration case studies were from low-income countries. It is likely that the trend in tropical countries is still firmly in degradation rather than restoration mode.

Recommendation

Recognizing and communicating ecosystem services, including distinguishing different types of services and translating these into, particularly local, benefits is an increasingly used component of ecosystem and biodiversity assessment. Ensuring that methods are transparent and that relative costs can be made with reasonable confidence is essential for a conservation agenda. Communicating the monetary or other value of tropical aquatic ecosystems is particularly important for informing decisions at local and national scales.

Topic 3: Institutional frameworks and stakeholders

A general disconnect in many parts of the world is the translation of national or international policies to local action (Egoh *et al*., 2012). The connectivity between national and local levels typically transcends through several tiers of government, involving a variety of government agencies and other stakeholders. The effectiveness with which a country manages its environment depends in part on the production and allocation of human and financial resources to run the management process. The model employed with some success in developed countries has relied on technically competent scientists producing bio-ecological information, reported and acted upon to develop and implement policy. The current state of the art of this process is following a cycle of adaptive management, with periodic reviews of monitoring and quality assurance, and adaptation as needed (Pahl-Wostl *et al*., 2007). Within this framework, cost-effective monitoring is essential, but while the value of long-term monitoring is well known (Lovett *et al*., 2007), in many parts of the tropics hydrological monitoring has declined markedly over the last five decades (Houghton-Carr and Fry, 2006). Collecting representative data of overall ecosystem quality needs both competent field workers, and effective institutional structures. In many countries this idealized management cycle is poorly constructed, even when underpinned by legal requirements, and connecting conservation needs to effective action is often sub-optimal, not well aligned with other policies, or lacking the required range of disciplines. This challenge is generally more pronounced in the tropics, and one of several reasons why the northern model may not function well in tropical developing regions.Primarily these relate to a lack of finances and ill-functioning institutional structures, which together restrict all other aspects of the management cycle.

That government-run, top-down environmental management agencies can only perform poorly when they lack financial resources and trained personnel has, in part, led to a tendency to develop more participatory management structures. Successful shared management such as that of benthic coastal resources in South America and Oceania (Richmond *et al.*, 2007), provides examples that can be applied elsewhere. New participatory institutional arrangements in many cases can complement local authorities' work and regulation, but not completely substitute governmental capacity for monitoring, oversight and management. Nevertheless, a contemporary view of aquatic conservation must, inevitably, be seen through the lens of local economies and their stakeholders. As such, arguments for protecting biodiversity per se, aquatic or otherwise, have limited appeal at either local or government levels (Wishart *et al.*, 2000). Linking policies to action requires recognition that aquatic conservation is of societal importance. In some cases, the use of economic valuation arguments (see above) may help, but this can only, at best, be one component. In the messy world of complex biology and social order, conservation needs to engage with a spectrum of stakeholders. Calls for stakeholder involvement in conservation and natural resource management often lack meaningful enagagement, and this remains a major and under-resourced challenge. Limited capacity to deal with increasing pressures on aquatic systems is common in the tropics, and high rates of illiteracy require alternative means for communicating with local communities. Low levels of formal education among the general population restricts communication of civil society with management agencies. Across tiers of decision-making, a shortage of trained professionals restricts the dialogue, and hence the design and implementation of conservation measures. Furthermore, in northern and temperate zones, the time and techniques needed to engage with stakeholders in natural resource management is generally grossly underestimated (Norgaard and Baer, 2005). Working across different cultural and educational settings in the tropics defies effective conservation action without the appropriate level of thoughtful planning and actions necessary for the slow process of social learning (Dewulf *et al.*, 2005). Achieving a change in conservation management requires cross-sectoral dialogue and meaningful engagement with local communities, business and different and hierarchical tiers of government. At all scales, from local to global, this is a highly complex discussion because the central components of conservation are landscapes and their people.

Developing human capital and skills from govererment to communities involves political decisions for prioritizing resources. While there are legitimate debates on provision and mechanisms for external support by donors, or international fiscal policies, the operational need for aquatic conservation ultimately requires national vision and investment. External financing can, and should, support conservation of some areas, but not the overall ecosystem service insurance for a nation. More project-orientated activities such as sporadically distributed research or training scholarships, or capacity building workshops can build awareness among relevant institutions but, given the scale and multifactorial nature of functioning ecosystems, can only be of limited benefit or act as a catalyst for building competence. The use of the 'stakeholder workshop' led by external professionals cannot realistically expect to achieve much unless carefully embedded in a local, and supported, process. Whatever the mechanism for building capacity, it requires a long-term vision that anticipates and plans for the required technical and relational skills and competencies within a range of institutions. Reliance on donor-led conservation is a fragile and probably ineffective solution. Building on the momentum of the SDGs, there is recognition throughout the tropics of the need for developing capacity across government institutions and civil society. A series of core principles for this are suggested by Sustainable Development net (www.SDplanNet.org), an internationally supported network for capacity development among government staff across and connecting national, sub-national and local institutions. These are: (1) multi-stakeholder processes and institutions; (2) integrated planning for vertical collaboration at different levels of government; and (3) scaling up implementation through cross-cutting policies providing multiple synergies.

Recommendation

Identifying and involving the tiers of government and stakeholders that influence conservation of habitats and species is an essential first step to effective management. Targeting capacity development to those who can make a difference for aquatic conservation is fundamentally important. Capacity development for individual competencies requires commensurately functional institutions. Coordinating institutional needs saves limited resources and creates a network of decision makers that can work together for the preservation of aquatic habitats. Skills needed for conservation management need to include relational as well as technical ones.

Topic 4: Monitoring, reporting and accountability

There is an old adage that 'you can't manage what you don't measure'. While this may be overstated where there is good general understanding of the effects of human pressures on aquatic health, learnt from similar well-studied areas, the lack of basic monitoring in many parts of the tropics severely hinders, or actively prevents water management or recognition of causal relationships. Where monitoring does occur, it is of little value without formal reporting and accountability. Recently Lu *et al.* (2015) called for five priorities for achieving the UN Sustainable Development Goals (SDGs). These included devising effective metrics, monitoring systems, and the standardization and verification of data. These apply as much to conservation needs and the meeting of the Aichi Biodiversity targets as they do to the SDGs. Techniques for monitoring aquatic ecosystems are well developed, both for assessing pollutants, or for sampling bio-indicators of pollution or other disturbance. What requires considerable and urgent development is locally applicable field guides and taxonomic keys, and the biologists who can use them. Conservation planning for targeting where sampling is most useful for aquatic conservation has been well developed in the last decade, supported by powerful and increasingly available geo-referencing and remote sensing technology (Linke *et al.*, 2007; Nel *et al.*, 2011). The challenge for many tropical countries is providing the necessary infrastructure and administrative capabilities that enable application of the technology at the appropriate local scales.

Connecting the technological know-how with the capacity at the field scale can be supported by two crucial, and entirely achievable, mechanisms. The first is the development of key demonstration activities and locally accessible field stations. A recent survey of the global distribution of biological field stations highlights a severe deficit of these in the tropics (Tydecks *et al.*, 2016). Strategically focused facilities provide the opportunities to act as regional hubs of data acquisition and training that, through links with cloud-based Spatial Data Infrastructure (SDI), provide access to regional datasets, as well as stimulating the depositing of those data from the various monitoring and research projects. Once the wider value of such a facility is demonstrated, it has high probability of both self-financing and crucial regional political support.

The other monitoring opportunity that is increasingly feasible, and needed, is the use of locally empowered communities to monitor their own resources (Aceves-Bueno *et al.*, 2015). Simple techniques such as the mini Stream Assessment Scoring System (www.groundtruth.co.za/projects/minisass.html), now widely used in South Africa, can be used to provide simple, but valuable, monitoring and promote community driven stewardship. While the ideas of citizen science are not new, the earlier vision of ecologists such as Carlson (1977), that mobilized citizens across the US for an annual Secchi 'dip-in' that continues to this day, are now becoming much more widespread through global mobile phone use.

The mechanics of conservation in the tropics need not only effective capacity development but also requires the support of citizens. While techniques such as mini-SAS are being rolled out, a major challenge is building awareness for conservation among an increasingly urbanized world, with most people in the tropics predicted to be living in cities by 2050 (IOM, 2015). Increasing urban populations not only have direct impact on aquatic conservation (Mcdonald *et al.*, 2008) but risk disconnection from the natural environment (Miller, 2005; Restall and Conrad, 2015). Actively bringing environmental education and advocating sustainability science to all citizens may no longer be a nicety, but an essential educational need. Linking that to open and green space within cities may not only help re-connect humans with their ecological dependency, and its demonstrable, social and health benefits, but engage urban voters in the support of a sustainable environment on which they ultimately depend.

Recommendation

Establishing well-functioning monitoring networks is a basic need for conservation management. Geospatial techniques that support catchment management and conservation planning are increasingly available. Field data needs effective quality assurance, and transparent reporting and reflection. Technical tools and skills development can be complemented by greater engagement with local communities, including their empowerment to monitor their own natural resources. This can build stakeholder confidence and common purpose within communities. More than anything, working with local communities, tropical or otherwise, takes time, patience and social skills.

FINAL COMMENTS

Many of the key questions in aquatic tropical ecology and conservation remain unanswered, hindering some of the world's most serious conservation problems. Ultimately, conserving aquatic, or other, biodiversity and habitats depends on whether they are considered necessary societal goals at both national and local levels. If so, then there is a serious need for better policy, monitoring and capacity development, and the societal and political awareness to meet the increasing challenges that will inevitably present themselves. It will require a vision of shared goals for economic development, sustainability and human well-being. A widespread congruence between aquatic and terrestrial biodiversity offers opportunities for greater alliance, and cost-effective strategies between aquatic and terrestrial conservation (Abell *et al.*, 2011; Flitcroft *et al.*, 2016). Nevertheless, conservation planning and action will need to embrace better an integrated approach across both habitat and institutional boundaries (Nel *et al.*, 2009b). The social setting is as important as the biological (Hunter, 2002). The tradition of a stringent protected areas approach has shifted in response to greater equity of resource use among local populations, and their future management for people as well as wildlife will increasingly be open to critical assessment. The gross pollution that afflicted many industrialized nations in the 1960s has generally been remedied, so a similar trajectory is possible for tropical developing countries. However, the authors or readers of this or other conservation orientated journals will not effect change to redress the decline of quality and extent of tropical waters unless they become more active in engaging with a spectrum of activities that may lie well outside their specific comfort zones. Otherwise, locally trained biologists, where they exist, will merely be the future recorders of a diminishing return.

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