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Achieving Aichi Biodiversity Target 11 to improve the performance of protected areas and conserve freshwater biodiversity

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ABSTRACT

1. The Strategic Plan for Biodiversity (2011–2020), adopted at the 10th meeting of the Conference of the Parties to the Convention on Biological Diversity, sets 20 Aichi Biodiversity Targets to be met by 2020 to address biodiversity loss and ensure its sustainable and equitable use. Aichi Biodiversity Target 11 describes what an improved conservation network would look like for marine, terrestrial and inland water areas, including freshwater ecosystems.

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2. To date, there is no comprehensive assessment of what needs to be achieved to meet Target 11 for freshwater biodiversity. Reports on implementation often fail to consider explicitly freshwater ecosystem processes and habitats, the pressures upon them, and therefore the full range of requirements and actions needed to sustain them.

3. Here the current progress and key gaps for meeting Aichi Target 11 are assessed by exploring the implications of each of its clauses for freshwater biodiversity.

4. Concerted action on Aichi Biodiversity Target 11 for freshwater biodiversity by 2020 is required in a number of areas: a robust baseline is needed for each of the clauses described here at national and global scales; designation of new protected areas or expansion of existing protected areas to cover known areas of importance for biodiversity and ecosystem services, and a representative sample of biodiversity; use of Other Effective Area-Based Conservation Measures (OECMs) in places where designating a protected area is not appropriate; and promoting and implementing better management strategies for fresh water in protected areas that consider its inherent connectivity, contextual vulnerability, and required human and technical capacity.

5. Considering the specific requirements of freshwater systems through Aichi Biodiversity Target 11 has long-term value to the Sustainable Development Goals discussions and global conservation policy agenda into the coming decades.

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INTRODUCTION

In 2010 in Aichi, Japan, 193 parties to the Convention on Biological Diversity (CBD) agreed to the Strategic Plan for Biodiversity (2011–2020) to address biodiversity loss and ensure its sustainable and equitable use. To achieve these aims, the plan sets 20 Aichi Biodiversity Targets to be met mostly by 2020 (CBD, 2010). Aichi Target 11 reads:

> By 2020, at least 17 per cent of terrestrial and inland water areas and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscape and seascape.

However, reports on implementation of Target 11 usually fail to consider explicitly freshwater ecosystem processes and habitats, the pressures upon them, and therefore the full range of requirements and actions needed to sustain them (Pittock *et al.*, 2015).

By volume, about 26% of the earth's inland waters are saline continental water ecosystems, and 73% are fresh water (Mittermeier *et al.*, 2010: 16–17). These freshwater ecosystems occupy less than 1% of the earth's surface and yet contain perhaps as much as 12% of all known species, including a third of all vertebrate species (Garcia-Moreno *et al.*, 2014). Freshwater ecosystems and their resources support most life on earth, sustain the livelihoods of billions of people, and provide a range of different goods and services that underpin our economies (Russi *et al.*, 2013). Paradoxically, the increased demand for fresh water is rated as the worst of the 'Global Risks in Terms of Impact' over the next 10 years to political, social and economic security, requiring difficult choices on allocation among users (World Economic Forum, 2015). Moreover, freshwater ecosystems are among the most threatened and degraded on Earth, suffering from key pressures including water abstraction, water pollution, destruction or degradation of habitat, flow modification, overharvesting of species, and invasive alien species (Garcia-Moreno *et al.*, 2014). Inland wetlands (which include brackish and saline systems) have declined in extent by 64–71% during the 20th century (Davidson, 2014).

Despite the importance of freshwater biodiversity (and the components of the ecosystems that support it) for nature and people, and their high level of threat, protected area networks historically have been established for terrestrial conservation (Allan *et al.*, 2010; Herbert *et al.*, 2010). Acknowledging this, IUCN (2014) recommended that 'Global protected areas should include a specific focus on coverage and management of freshwater ecosystems in their own right rather than as a component of terrestrial systems…'.

Here, current progress on achievement of Target 11 for freshwater biodiversity is assessed. While saline systems are ecologically important, the focus is principally on freshwater ecosystems owing to their high level of threat, ecological importance, and key role in people's livelihoods. Following a similar approach by Woodley *et al.* (2012), here the Target is split into eight clauses, to explore what each of these mean for freshwater biodiversity, what their current status of implementation is, and what the key gaps are. Finally, a set of recommendations for moving forward on achieving Target 11 for freshwater biodiversity by 2020 is proposed.

1. '… 17 per cent of terrestrial and inland water areas and 10 per cent of coastal and marine areas…'

Assessing the coverage of freshwater protected areas

Inland water areas and terrestrial areas are both included within Target 11's call for 17% coverage, and are usually reported together, making it difficult to differentiate between them and, therefore, assess protected area coverage of inland water ecosystems specifically. The diverse definitions of 'inland water' and even 'wetlands' further increases the complexity of reporting progress towards this particular aspect of the target. The CBD uses the Ramsar Convention's definition of wetlands which is broad and includes artificial wetlands such as ponds and reservoirs, brackish inland waters, and areas of marine habitat to a depth of 6 m below low tide level (Ramsar Convention Secretariat, 2013). Thus, when assessing the coverage of protected areas of inland water ecosystems, the scope of such assessment, and the definition of inland waters considered, needs to be clear.

An assessment of protected areas coverage of wetlands using the World Database on Protected Areas (WDPA; IUCN and UNEP-WCMC, 2015) and the Global Lakes and Wetlands Database (GLWD; Lehner and Döll, 2004) showed that 20.7% of the total area of the nine major inland water types included in the GLWD are within protected areas (Juffe-Bignoli *et al.*, 2014: p. 16). This suggests the 17% mark has been achieved in terms of the inclusion of inland water areas. However, some of these protected areas may already be seriously compromised by their design and location. Although a region is defined as a protected area, it might still be affected by drivers of threats that originate outside the protected area, with consequences that cannot, therefore, be easily controlled within the protected area (see also section re clause 3). For instance, this analysis does not consider the propagation of threats from upstream unprotected areas which are fundamental to ensuring effective protection of highly connected systems such as rivers (Pittock *et al.*, 2015). For example, within the Iberian Peninsula the European Natura 2000 network includes many protected areas that are subject to threats from upstream regions (Hermoso *et al.*, 2015a). According to Holland *et al.* (2012) although 19.3% of the total river lengths in Africa are covered by protected areas, this includes rivers acting as boundaries for protected areas, which are unlikely to be effectively protected (Abell *et al.*, 2007; Nel *et al.*, 2007; Darwall *et al.*, 2011).

The designation of sites through the Ramsar Convention has led to the most extensive list of sites focusing on wetland conservation. The Ramsar Secretariat reports 2208 Ramsar sites covering 2.1 million km^2 (Ramsar, 2015a), although many of these are brackish, coastal or entirely marine sites and the distribution of freshwater sites across biogeographical and wetland types is not representative of the known wetland distribution. Nevertheless, according to Pittock *et al.* (2015), in February 2014 there were 795 inland sites that were exclusively freshwater wetlands on the Ramsar List, covering a total area of 1 million km^2.

Conclusions and recommendations

The percentage of freshwater areas that are covered by protected areas should be measured independently of the terrestrial regions, to ensure that freshwater ecosystems are adequately represented. However, there is still no comprehensive and consolidated assessment on how well inland waters are covered by protected areas. This is due in part to the difficulty in defining the full extent of freshwater ecosystems (e.g. how far beyond the immediate riparian habitat does one extend), and to the different ways in which wetlands have been defined (see above). Completing this assessment is important for building a baseline upon which progress toward Target 11 can be measured. More importantly, 17% and 10% are politically negotiated values and not scientifically defined end points (Woodley *et al.*, 2012). A considerably higher proportion of area will need to be protected to achieve, for example, an ecologically representative network (Butchart *et al.*, 2015); there are different strategies to achieve this efficiently (Venter *et al.*, 2014; Montesino-Pouzols *et al.*, 2014), although none of these studies have specifically focused on assessing the needs for freshwater biodiversity at a global level. More importantly, a global network of protected areas (and OECMs; see section re clause 7) must fulfil other equally important qualitative requirements. These are covered in the remaining clauses of Target 11 discussed below.

2. '…especially areas of particular importance for biodiversity and ecosystem services…'

This clause refers to the need to conserve sites that are important because of the biodiversity they support and/or the ecosystem services that they provide. The existing protected area network is biased towards places that are at higher elevation, have steeper slopes, and are further from roads and cities (Joppa and Pfaff, 2009) rather than other locations of particular importance for biodiversity (Butchart *et al.*, 2012). Even so, the existing protected area network does not adequately cover the upper parts of many river catchments (Garcia-Moreno *et al.*, 2014). Conversely, for example, Hermoso *et al.* (2015a) noted that in the Iberian Peninsula the lowland areas are often inadequately covered by the Natura 2000 network of reserves. These lowland areas are important for the conservation of many species, and provide a significant conservation challenge because they are extensively modified for other land uses such as agriculture and urban development.

Areas of importance for biodiversity

Key Biodiversity Areas (KBAs; Langhammer *et al.*, 2007) have been used to assess the extent to which protected areas are covering areas of importance for biodiversity (Butchart *et al.*, 2012, 2015). To date, only two networks of KBAs have been identified worldwide: Important Bird and Biodiversity Areas (IBAs: >12 800 sites of global avian significance: BirdLife International, 2014) and Alliance for Zero Extinction sites (AZEs: 587 sites holding the last remaining population of one or more highly threatened species: Ricketts *et al.* 2005). Among the 6369 IBAs identified for inland water species, protected areas (as documented in the January 2013 version of the WDPA) cover 21% of sites completely, and 45% partially; 34% of sites were entirely unprotected. On average, just 42% of each site is covered by protected areas (analysis of data in Butchart *et al.*, 2015). For 236 AZEs identified for mammal, bird and amphibian species that use inland waters, 22% are completely covered, 37% are partially covered and 41% have no coverage by protected areas. On average 42% of their area is covered. These results combined demonstrate there is a considerable shortfall in the degree to which protected areas cover areas of particular importance for freshwater biodiversity. IUCN is currently convening a process to develop a standardized set of overarching criteria to identify KBAs (IUCN, 2015), building on and encompassing IBAs and AZEs, as well as other KBAs identified in freshwater ecosystems in more than 90 countries (Foster *et al.*, 2012; Holland *et al.*, 2012).

In addition to the KBA approach, advances in systematic freshwater conservation planning (SCP) over the last two decades provide a solid set of additional fundamental principles (Higgins, 2003), and a variety of tools (Moilanen *et al.*, 2008; Kingsford and Biggs, 2011), for identifying areas of importance for biodiversity and ecosystem services in a representative, comprehensive and spatially efficient manner, and for setting priorities for conservation (Thieme *et al.*, 2007; Hermoso *et al.*, 2009; Khoury *et al.*, 2011). There is also a growing literature with examples of freshwater conservation planning (Thieme *et al.*, 2007; Hermoso *et al.*, 2009). Nevertheless, one of the challenges ahead is to translate these plans to on-the-ground action (Roux *et al.*, 2008; Barmuta *et al.*, 2011). The new KBA standard's criterion E (IUCN, 2015) for sites identified as highly irreplaceable using systematic conservation planning methods aims to draw from the strengths of both KBA and SCP approaches (Di Marco *et al.*, 2015).

Areas of importance for ecosystem services

Protected areas have an important role in the downstream supply of fresh water for ecosystem functioning and delivery of services for human communities (Dudley and Stolton, 2003). Despite many local case-studies and national scale analyses, there is only one, preliminary, global assessment of the degree to which protected areas cover areas of importance for freshwater ecosystem services (Harrison *et al.*, 2016). Even this study was restricted to assessing water provision (and threats to this service), rather than other provisioning or regulating hydrological services. In a more regionally detailed study, Darwall *et al.* (2011) showed that areas of highest freshwater species richness and threat in Africa are congruent with areas where people have high reliance on ecosystem services. However, many of the freshwater ecosystems in Africa are not protected (see section re clause 1) and even where freshwater habitats are included within protected areas, they are still threatened by upstream pressures and poor management (Hermoso *et al.*, 2015a).

Conclusions and recommendations

More research is needed to understand better the degree to which protected areas cover inland water areas of particular importance for biodiversity and ecosystem services. It will be important to complete the identification, globally, of KBAs including sites identified through systematic conservation planning under KBA criterion E, covering a wider range of freshwater taxa and elements of biodiversity (e.g. threatened freshwater ecosystems). This requires a more complete knowledge of the distribution of species and their extinction risk, through assessments for the IUCN Red List of Threatened Species (Carrizo *et al.*, 2013; Collen *et al.* 2014). Similarly, to understand better the extent to which protected areas cover inland water areas of high importance for ecosystem services there is a need for further studies to define the relationship between ecosystem structure and function (Cardinale *et al.*, 2012; Tomimatsu *et al.* 2013) and quantify provision of ecosystem services from freshwater ecosystems.

Achieving this clause of Target 11 will therefore require strategic expansion of national protected area networks, targeting initially those sites already known to be important for freshwater biodiversity and service provisioning, while recognizing the contribution that their protection would add to other clauses of Target 11 (Butchart *et al.*, 2015) and also to other Aichi Biodiversity Targets (Butchart *et al.*, 2012; Venter *et al.*, 2014; Di Marco *et al.*, 2015).

3. '…effectively managed…'

Protected area management effectiveness (PAME) is described within IUCN best practice as comprising six elements within the management cycle: context, planning, inputs, processes, outputs and outcomes (Hockings *et al.*, 2006). Several tools for protected area management exist and, recently, a simple tracking tool for management effectiveness specific to wetlands was endorsed by the Ramsar Convention for use in tracking management across Ramsar sites (Resolution XII.15; Ramsar, 2015b). The Ramsar tool highlights many of the issues that are necessary to consider when assessing the effectiveness of a protected area for conserving freshwater values.

Knowledge gaps, status, threats and management needs of fresh water

Most protected area managers are terrestrially focused, such that some protected area networks lack the knowledge, skills, and funding to adequately assess and define appropriate management for freshwater ecosystems (Thieme *et al.*, 2012). In January 2016, only about 52% (856) of Ramsar sites with freshwater wetlands had management plans and less than half (42%) of National Reports (146) by Ramsar Convention Contracting Parties prior to COP XII (2015) reported on their effectiveness for conserving wetland values. Although there are some tools or guidelines for the integrated assessment of the status and management of wetland ecosystems (Springate-Baginski *et al.*, 2009; Bunting *et al.*, 2013), a consistent and robust framework has not been adopted universally by protected area managers. Pittock *et al.* (2010) provide examples from five wetland sites in the Murray Basin where a lack of understanding resulted in bad management decisions for freshwater ecosystems.

As climatic and hydrological cycles are intimately linked, management effectiveness tools must also address climate change adaptation for freshwater protected areas, covering, at a catchment scale, such measures as environmental flows, re-operation of water infrastructure, habitat restoration for resilience and conservation of climatic refugia (Lukasiewicz *et al.*, 2013; Pittock *et al.*, 2015). In addition, Ostfeld *et al.* (2012) have demonstrated, through case studies, the importance of engaging relevant stakeholders in building plans for environmental flows and decision-making tools for climate change, in order to develop strong ecosystem and water management practices. In many cases, there is also inadequate knowledge available of the freshwater species and systems that occur within a protected area and the risks to them from beyond the protected areas, as well as insufficient resources for comprehensive surveys. In these situations it will be necessary to use the best available data and surrogates (Tisseuil *et al.*, 2013) for assessing management effectiveness.

Mismatch between protected area boundaries and hydrological units

Rivers, lakes, and wetlands collect and convey water and materials flowing across the land and emanating from groundwater sources. Lateral and longitudinal connectivity is one of the defining characteristics of freshwater ecosystems. The highly connected nature of water and land, the linear network, and directional flow of freshwater systems make them particularly vulnerable. The potential for protected areas to function as freshwater ecosystem refugia and be effective is highly dependent on the broader context in which they are situated. Abell *et al.* (2007) suggest a tiered approach that includes 'focal sites' that sustain populations of rare and/or threatened species, where strict conservation measures are applied, embedded in 'critical management zones' (e.g. river reaches) that have a broader scheme of conservation and management opportunities to help protect the focal sites, and 'catchment management zones', that include focal sites and critical management zones, and are designed to support the multiple uses and needs of the catchment. Allan *et al.* (2010) illustrated schematically how this approach could integrate multiple different uses within a catchment. Hermoso *et al.* (2015c) showed how application of these methods (Abell *et al.*, 2007) would improve the effectiveness of conservation for freshwater biodiversity in the Natura 2000 reserve network in the Iberian Peninsula. Lukasiewicz *et al.* (2013) recommend identifying and prioritizing free-flowing rivers and more climatically resilient freshwater habitats, even if this requires restoring areas that are currently degraded.

Conclusions and recommendations

Approaches to assess management effectiveness of protected areas exist but, in the context of evaluating the implementation of Target 11, a robust framework has neither been consistently nor comprehensively applied. More work is therefore needed to: (1) update PAME tools to ensure that they assess effectiveness across all components of aquatic integrity and address climate change adaptation for freshwater protected areas; (2) encourage use of these tools for all protected areas that include inland waters; (3) design new protected areas and update existing ones to have the maximum possible potential to ensure effective conservation of freshwater ecosystems – for example, by using catchment boundaries instead of rivers, adaptively managing protected areas to enhance resilience to global changes (such as by identifying existing and future conservation corridors) and ensuring that elevational and latitudinal ecological gradients remain intact or are restored.

4. '…equitably managed…'

The equitable management of protected areas described in Target 11 incorporates the dimensions of distributive and procedural justice, including stakeholder recognition (Martin *et al.*, 2013). This means that both the distribution of benefits and burdens (distributive) as well as the decision-making processes and stakeholder engagement (procedural) ought to be equitable. However, there is no single one-size-fits-all solution or universal principle that defines what equitable management looks like (Ingram *et al.*, 2008), as equity will always be context-specific with conflicting interpretations and definitions. Incorporating equity into management (both in processes and outcomes) exposes a number of potential conflicts that must be considered by management: competing values (what should be done), temporal scales (long-term versus short-term impacts), inclusion (where stakeholders are recognized in process and distribution of burdens and benefits), spatial scale (what is equitable at the local level may not be equitable at a national or global scale) and history (how resource decisions of the past affect present stakeholders).

Water as a resource and its implications for equitable management

The nature of water as a resource poses additional complexities for implementing equity in protected areas. Neal *et al.* (2014 present four reasons why water is a unique resource with profound justice implications for its management: (1) the spatial and temporal uneven distribution of water; (2) the fact that water is essential for both human and non-human life; (3) the myriad goods and services provided by water that ensure human well-being; and (4) the ensuing political dimensions of power asymmetries affecting water governance.

The uneven distribution of water (both spatially and temporally) creates burdens and benefits to different stakeholders (Neal *et al.*, 2014). The equitable management of freshwater protected areas must consider not only present conflicts, but historical and inter-generational conflicts as well (West *et al.*, 2006). Similarly, the interconnectedness of freshwater systems means that management decisions have widespread spatial impacts (as discussed above). Water infrastructure, such as dams, has profound equity impacts (both positive and negative) on different upstream and downstream groups and these can change with time (Niasse, 2002).

The extent to which the rights to water of non-human species are reflected and given priority in management actions depends on whether intrinsic or extrinsic values are assigned to nature (Crawhall, 2015). While the 'right' of non-human life to water is not as universally established as the human right to water, which is widely recognized at the international level (Selborne, 2001), it is expanding as environmental water needs are increasingly recognized (see Godden, 2005, for examples in South Africa and Australia). The recognition of non-human rights affects the type of stakeholders and conservation actions deemed to be legitimate by including proxies for non-human species, such as environmental NGOs. It may also lead to the prioritization of some non-human species over others; for example, the preservation of an endangered species over an introduced one (Dell'Amore, 2013).

Water also provides numerous social, cultural and economic benefits that affect human well-being, and often a water body will serve multiple uses at the same time (Kumar *et al.*, 2011), and protected areas can have significant positive effects on human well-being (Stolton and Dudley, 2010). Depending on how they are established and managed, freshwater protected areas can either help secure goods for local people (e.g. by facilitating continuation of traditional fishing while preventing developments that would damage fish), or alternatively can reduce local access to ecosystem services, such as by banning fishing altogether. Which uses are given priority depends on recognition, the particular values being protected, the types of stakeholders included and their values. Fresh water is both a private good used as an agricultural and industrial resource and a public good supporting social and cultural values.

Conclusions and recommendations

This section has outlined the complexities of equitable management of freshwater protected areas. While there are no readily available criteria to evaluate equitable management as yet, this section has outlined the concepts and boundaries of equity that would need to be considered and adapted to individual contexts. Given the multitude of conflicting values around water and protected areas, the implementation of equitable management must concentrate on ensuring procedural equity through well-established best practice engagement, paying special attention to unequal power relations (Ramsar Convention Secretariat, 2010; Neal *et al.*, 2014; Dovers *et al.*, 2015). Yet, some important steps towards addressing distributive equity for ecosystem needs have recently been made. For example, Parties to the Ramsar Convention (Resolution XII.12; Ramsar, 2015b) recognized that 'ensuring the availability of the water required by wetlands will promote both their biodiversity and the sustainable use of their components, [and will] contribute to the sustainable management of water in agricultural areas, and maintain the impacts of the use of natural resources within ecological limits in order to guarantee biodiversity conservation.'

5. '…ecologically representative…'

Ecological representation as it relates to conservation is defined by Groves (2003) as, 'the need to represent occurrences of each community or ecosystem across the environmental gradient in which they occur in a system of conservation areas'. There are several challenges with ensuring representation of freshwater systems within protected areas to meet Target 11. First, freshwater systems have been described as hierarchically nested (Frissell *et al.*, 1986) such that no single level of the hierarchy can represent all elements. Second, the highly connected nature of freshwater systems (which may include fluvial, lacustrine, palustrine and estuary areas) makes them particularly vulnerable to fragmentation (Fagan, 2002), and upstream/downstream influences (Fausch *et al.*, 2002) (see sections re clauses 3 and 6), such that the ability of the species and ecosystems to persist needs to be accounted for in conjunction with ensuring adequate representation across the catchment (Dudgeon *et al.*, 2006; Nel *et al.*, 2009, 2011; Linke *et al.*, 2012). Third, freshwater ecosystems are highly dynamic and evolving systems (Stiassny *et al.*, 2010); for example, rivers change course and features, pools are filled with sediment to become marshes. All of these changes to ecosystems and species communities are further modified by the effect of climate change (Parmesan, 2006). Thus, protection of sites, aimed at the conservation of ecologically important freshwater areas now and into the future, must allow for the capacity of ecosystems to adjust, and particularly to keep in step with climate.

Assessing 'ecological representation'

Four levels of organization common in scientific literature may be used when assessing biodiversity: species, species assemblages, ecosystems, and ecoregions. Each of these levels captures different elements of freshwater biodiversity, with none of them able to represent all biological or ecological characteristics. At the species level, ecologically representative characteristics include genetic diversity (traits); phenotypic variability (behaviour); and patterns of distribution or abundance, including the relationship between species and habitat types. These characteristics of species-level biodiversity are well known for only a small number of species, often at regional rather than global extents. Conservation of individual freshwater species may encompass entire ecosystems, and catchments, especially for long-range migratory species. However, protection needs for one species may not necessarily be optimum for all species. Aspects of species assemblages that must be accounted for in achieving ecologically representative freshwater protected areas include species composition, trophic structure, relative abundance, and their interaction (such as predator–prey dynamics) (Townsend, 1989). Ecosystem-level characteristics under consideration can include species composition and diversity, hydrogeomorphic and connectivity characteristics, and species assemblage dynamics (Vannote *et al.*, 1980; Frissell *et al.*, 1986; Fausch *et al.*, 2002). Assessments of ecosystem diversity (including environmental processes that sustain the system) at the extent of catchments or whole basins may inform comprehensive conservation and management plans (Higgins *et al.*, 2005; Thieme *et al.*, 2007; Heiner *et al.*, 2011). The identification of locally significant species, processes, and ecosystems within the broader context of an ecoregion or basin can guide the identification of additional protected areas (Higgins *et al.*, 2005; Higgins and Duigan, 2009; Khoury *et al.*, 2011). The ecosystem scale may be used to capture patterns of river connectivity that are critical in providing the freshwater, nutrient inputs, and habitat diversity necessary for the persistence of aquatic species (Nel *et al.*, 2011). At broad spatial extents, freshwater ecoregions (based on unique regional species composition) capture patterns of biodiversity that may be assessed at a global scale (Abell *et al.*, 2008).

Conclusions and recommendations

A comprehensive global assessment of the adequacy of protection of freshwater species and systems is currently lacking. However, where gap assessments have been completed, they show a pattern of under- or uneven representation of freshwater ecosystems and species. For example, 18% of imperilled freshwater fish occur within national parks in the US (Lawrence *et al.*, 2011); only 16 of the 112 main river ecosystems in South Africa are moderately to well represented within protected areas (Nel *et al.*, 2007); and aquatic features of Michigan's inland waters are unevenly represented within the state's network of protected areas (Herbert *et al.*, 2010). In order to address the lack of a global gap assessment, expansion of the assessments of freshwater species on the IUCN Red List (Carrizo *et al.*, 2013; Collen *et al.*, 2014) and the completion of a spatially explicit global classification of freshwater ecosystems are necessary.

In the face of climate change, the resilience of protected areas to absorb changes, or to make the transition into new ecosystems, is a concern (Beaumont *et al.*, 2011). Sustaining biodiversity with a changing climate requires the capacity of systems to adjust. Metapopulation dynamics of colonization and recolonization require conservation of environmental processes and habitats — including climate refugia — and maintaining and enhancing connectivity (Rieman and Dunham, 2009). Analysis of ecological representativeness at each level of organization can inform local, regional, and global planning in a changing climate.

6. '…well-connected systems…'

When designing efficient and effective conservation area networks for freshwater biodiversity the longitudinal, lateral and surface/groundwater connections that define these systems need to be taken into account (Fausch *et al.*, 2002; Turak and Linke, 2011). Longitudinal connectivity allows long- and short-distance migrations of biota through river networks and is important for dispersal, reproduction and long-term population dynamics of many species of fishes. Lateral connectivity between the river channel and aquatic habitats (e.g. lakes and wetlands) on the adjacent floodplain is important to maintain the exchange of nutrients (Ward, 1989), and to maintain viable populations that depend on the floodplain for food and habitat during early life stages, and use the river channel as a dry season refuge (Welcomme, 1979). Vertical connectivity is also crucial given the dependence of some surface ecosystems on groundwater contribution (Ward, 1989). Planning for connectivity is especially important for adaptation to ecosystem change as a result of climate change (see previous section on clause 5).

There are methods available to account for the specific needs of freshwater systems in conservation planning, such as longitudinal (Linke *et al.*, 2007; Moilanen *et al.*, 2008; Hermoso *et al.*, 2011) and lateral (Hermoso *et al.*, 2012) connectivity. However, these methods have been used rarely to inform on-the-ground implementation of conservation. For example, Roux *et al.* (2008) used systematic planning methods to increase the soundness of existing protected areas for freshwater biodiversity, proposing a theoretical redesign of Kruger National Park. Hermoso *et al*. (2015b) identified a minimum set of priority areas that should be added to the Natura 2000 reserve network in the Iberian Peninsula, with longitudinal connectivity of reserves being a key factor in identifying new priorities. Although both of these are academic exercises, they demonstrate how scientific advances that incorporate the connectivity of freshwater networks can be used to enhance the capacity of protected areas for freshwater conservation in a cost-effective way.

Conclusions and recommendations

Further effort is required in the near future to improve the capacity to assess the effective connectivity of rivers worldwide and integrate this information in conservation assessments. The recently developed HydroBASINS database (Lehner and Grill, 2013) provides spatial information on sub-catchments and associated river networks for the whole world, including information on flow distances, and discharge and flow regimes, which is useful for assessing connectivity of rivers. This can be combined with data such as the Global Reservoir and Dam (GRanD) Database (Lehner *et al.*, 2011) and King's College London Global Dams Database (Mulligan *et al.*, 2009), to identify areas of fragmentation. In ongoing work, WWF and partners are developing an updated global registry of the status of connectivity of river systems (WWF, 2015), that can be tracked over time and help to inform the location of free-flowing rivers. This will be a valuable input to comprehensive river basin planning. Existing tools, and information developed in the coming years, should then be used to identify additional areas that would enhance the adequacy of protected areas for freshwater conservation. Forward planning to ensure ongoing connectivity of freshwater ecosystems within and between protected areas, via conservation corridors of rivers, lakes and wetlands across catchments, will be a particularly important component of adaptive management addressing landscape scale changes induced by climate change and other anthropogenic effects. Creating conservation networks that include all spatial needs important for freshwater systems could potentially result in large areas (Hermoso *et al.*, 2015). This could compromise the implementation of conservation recommendations for freshwater biodiversity under traditional conservation schemes of strict protection. However, as noted in the section re clause 3, alternative schedules have been proposed to make the implementation of conservation more flexible, following the tiered and multi-zoning structure of different management regimes (Abell *et al.*, 2007). The implementation of this approach is still in its infancy (Thieme *et al.*, 2007; Hermoso *et al.*, 2015c) but should receive more attention in the coming years.

7. '...other effective area-based conservation measures...'

'Other effective area-based conservation measures' (OECMs or conserved areas) can play a key role in achieving Target 11. However, five years since the Target was adopted, the continuing effort invested in developing guidance for protected areas (including Borrini-Feyerabend *et al.*, 2013; Worboys *et al.*, 2014) has not been matched by a similar focus on OECMs (Jonas and Lucas, 2013) and there is no globally agreed definition of OECMs, so their extent cannot be quantified (Bertzky *et al.*, 2012; Woodley *et al.*, 2012; Juffe-Bignoli *et al.*, 2014).

Unofficial guidance has been developed by IUCN members, with a consensus that OECMs should be those areas that are largely equivalent to protected areas but not officially recognized as such by governments (Lopoukhine and de Souza Dias, 2012; Woodley *et al.*, 2012). Since then, a broader set of questions about the exact nature of OECMs has been posed (Jonas *et al.*, 2014) and an IUCN World Commission on Protected Areas Task Force has been established to explore the issue further. Yet while the exact definition of OECMs remains uncertain at present, it can reasonably be expected that inland wetlands that are not protected areas but nevertheless effectively conserve biodiversity in the long term will be given greater future recognition for their contribution to Target 11. Two instances of this occurring in freshwater systems include 'secondary voluntary conservation' (where conservation may be a desired result of governance as a secondary, implicit or not fully conscious, objective), and 'ancillary conservation' (when conservation is a fully unintended consequence of managing nature), both with a reasonable expectation to be maintained in the long term (Borrini-Feyerabend and Hill, 2015). An example of the latter is the Murray Wetlands Working Group's work with farmers on the floodplain of the River Murray in Australia. Although there are major wetland nature reserves along the river, they are tens of kilometres apart and mainly comprise large river red gum forests on the lower elevation floodplains (Pittock *et al.*, 2010). Smaller wetland ecosystems may form 'stepping stones' among these major wetland reserves and are habitats for different elements of freshwater biodiversity. To conserve such sites that are cut off from natural water flows in the agricultural landscape the Working Group (from 2001 to 2008) partnered with irrigation water supply companies and 136 farmers to provide 20 044 ML of water to fill 162 privately owned wetlands covering 3368 ha (MDWWG, 2016). While the participating farmers were selected because of their commitment to conservation and verbally agreed to maintain the restored wetlands, no legally binding agreement preserves these sites. The long-term conservation effectiveness of these area-based measures lies in the social institutions established among the land owners and supporting organizations.

Notwithstanding current proposals, agreeing what constitutes an OECM – and how the effectiveness of OECMs might be guaranteed and measured over time – invokes technical and political challenges that will take time and effort to resolve. The River Murray wetlands therefore provide one example for testing ideas about OECMs that can be applied more widely, not only in other freshwater sites but also in terrestrial and marine environments.

Conclusions and recommendations

The role of OECMs is important in meeting Target 11 given that it is unlikely that sufficient expansion of formal protected areas will be feasible by 2020 to meet the multiple components of Target 11 (Butchart *et al.*, 2015), and given the need to manage for representative conservation areas of freshwater ecosystems that are a focus of human settlement and use. The present protected area coverage for freshwater biodiversity is insufficient, and does not take into account the need for upstream and downstream protection. The importance of these effects in inland water systems means that strong connectivity is especially vital for freshwater ecosystems (see section re clause 6). OECMs, in their various forms, sizes and management approaches, can play a key role in facilitating connectivity between protected areas.

8. '…integrated into the wider landscape and seascape.'

Actions towards integrating protected areas into the wider landscape and seascape have generally focused on establishing connectivity among protected areas to facilitate species movement and persistence of populations (Woodley *et al.*, 2012). For fresh waters, the main focus has been on enabling migration within river basins and ensuring good water quality in rivers upstream and downstream from existing terrestrial protected areas (Convention on Biological Diversity, 2010; also see the section on clause 6). Where protected areas are truly integrated into the wider seascape and landscape these actions would be considered in a regional context, together with information about which components of freshwater biodiversity remain outside of protected areas and how important these components are for the persistence of regional biodiversity (Turak *et al.*, 2011).

Another important aspect of integration is the contribution of freshwater species and ecosystems to the conservation of marine and terrestrial ecosystems and species. While there is substantial understanding of interdependencies among species associated with the different realms, especially at the interfaces (Naiman and Decamps, 1997), conservation planning methods that link terrestrial, freshwater and marine realms are relatively recent (Beger *et al.*, 2010) and these still do not explicitly estimate the potential benefits of actions to different realms (Adams *et al.*, 2014). The steps proposed for estimating these benefits and integrating them into conservation plans include assessments of threats in each realm, construction of action response curves for actions under consideration, and the use of existing or purpose-built software to optimize the allocation of actions spatially (Adams *et al.*, 2014).

The place-based management strategies proposed by Abell *et al.* (2007; and see the section on clause 3) can be very helpful in developing conservation plans that have a high chance of implementation (Barmuta *et al.*, 2011) while enhancing integration of protected areas into the wider landscape and seascape. For example, freshwater focal sites may be confined to formal protected areas while critical management zones and catchment management zones could help determine where other effective area-based conservation measures (OECMs) may be implemented. Recent systematic approaches to freshwater conservation planning (Linke *et al.*, 2011) and in particular advances in integrating connectivity and refugia into these approaches (Hermoso *et al.*, 2013) can help delineate the different zones. Approaches that integrate contributions from different land uses across the entire landscape, allowing the evaluation of spatially explicit management scenarios (Turak *et al.*, 2011), can be particularly useful for linking freshwater conservation with the wider landscape and seascape. These approaches add the benefits of actions within protected areas to actions outside of protected areas, e.g. sustainable grazing management, and stabilization of actively eroding soils (Turak *et al.*, 2011). Many such ambitious systematic freshwater conservation planning exercises have so far failed to be implemented (Barmuta *et al.*, 2011). Linking protected area design and management with local and national planning is likely to increase the implementation of these plans and improve the integration of protected areas into the wider landscape and seascape. National Biodiversity Strategies and Action Plans (NBSAPs) were previously identified as potentially having a significant role in achieving the 'integration into the wider landscape and seascape' clause of Aichi Target 11 (Juffe-Bignoli *et al.*, 2014). A global assessment of how NBSAPs integrate protected areas into national planning does not exist (Leadley *et al.*, 2014) and there seems to be even less information on links with other relevant planning activities such as water resource use, agricultural and urban development, threatened species recovery plans and threat abatement plans.

Conclusions and recommendations

Implementing the framework proposed by Abell *et al.* (2007) will help the integration of freshwater biodiversity conservation into the wider landscape. This should be combined with methods for systematic freshwater conservation planning, especially those that enable cross-realm planning, account for ecosystem services, and combine protective and restorative actions inside and outside protected areas to optimize the persistence of regional biodiversity. These methods and tools should be considered for NBSAPs and local or regional planning processes including water resource planning and land-use planning. These larger planning processes should also align with and guide planning activities that directly lead to practical actions such as protected area management plans, threat abatement plans, and threatened species recovery plans.

DISCUSSION

Despite the fact that inland waters are mentioned in Target 11, and that they have an undeniable importance for people and nature (Vörösmarty *et al.*, 2010; Green *et al.*, 2015), little attention has been given to inland waters when reporting on the implementation of the Target (but see Leadley *et al.*, 2014). To address this apparent information gap, the CBD Target has been split into eight clauses and, focusing on their application to fresh water, some guidance is provided on how to interpret each of them, and their status of implementation, when appropriate information is available.

Bringing the Target's clauses together

Assessing the implementation of Aichi Biodiversity Target 11 requires much more than just measuring coverage (Woodley *et al.*, 2012; Juffe-Bignoli *et al.*, 2014). It requires the measurement of all other aspects of the Target and, in particular for fresh water, it requires careful accounting for the factors that are unique for freshwater ecosystems. For example, a superficial review of data (see the section on clause 1) suggests that the 17% coverage for freshwater ecosystems might have been met already. However, other components of the Target are far from being achieved (see above), and many protected areas covering inland waters are not designed and managed to conserve freshwater biodiversity, do not account for upstream threats, nor do they consider the dynamism and temporary aspects of many freshwater ecosystems. Earth observations and spatial modelling (MacKay *et al.*, 2009; Gardner *et al.*, 2015 and the example cited therein), global indicators such as the Wetland Extent Index which integrates data on wetland change from sites worldwide (Dixon *et al.*, 2016) or indicators that consider hydrological functioning and connectivity of wetland systems (Jeftic *et al.*, 2011), will be especially useful tools in achieving this.

Achieving a baseline to assess progress

One of the challenges in addressing clauses in Target 11 is that with the exception of the 17% value of coverage, the end point is not clearly defined for any of the clauses; hence it was not possible here to determine freshwater-specific endpoints. Having quantifiable definitions of success is important for measuring progress towards Aichi Biodiversity Targets in terms of distance to a defined end-point from a known baseline (Tittensor *et al.*, 2014). Efforts to build this baseline at a national and global scale should focus on five key actions: (1) identifying areas of importance for biodiversity and ecosystem services and assessing their protected area coverage; (2) assessing ecological representation of freshwater biodiversity in protected areas; (3) developing mechanisms to assess how well protected areas are being managed for fresh water (requiring comprehensive consultation with managers); (4) creating a database of large freshwater conservation initiatives that promote connectivity and integration in the wider landscape; and (5) agreeing a definition of OECMs, specific to fresh waters and identifying where OECMs are at present contributing further to the conservation of freshwater biodiversity and resources, or could do so in the future.

Key considerations to implement Aichi Biodiversity Target 11 for freshwater systems

Table 1 summarizes the recommended actions that will help ensure that freshwater ecosystems are better represented in actions towards Target 11. The next major task is to set priorities or to identify interdependencies and trade-offs between these different actions. This is of paramount importance but not within the scope of this paper.

One immediate contribution to the achievement of Target 11 for freshwater biodiversity would be the designation of new protected areas or the expansion of existing protected areas to provide a more extensive and better connected network that covers known areas of importance for biodiversity and ecosystem services, and representative samples of biodiversity. This is a large challenge, especially in the face of conflicting land-use needs and climate change (Thieme *et al.*, 2010; Ostfeld *et al.*, 2012). To overcome these challenges a comprehensive assessment of costs and opportunities is needed to identify priorities for action and present cogent social and economic arguments for protecting the ecosystem services contained in the candidate protected areas, and supporting arguments from the perspective of biodiversity conservation (Russi *et al.*, 2013; Green *et al.*, 2015). Systematic Conservation Planning tools can be helpful in this context.

The establishment of protected areas for freshwater ecosystems needs to be combined with improved management of existing protected areas (Thieme *et al.*, 2012). The Integrated Water Resources Management (IWRM) approach can

Table 1. Status of implementation of each clause of Aichi Biodiversity Target 11 for freshwater biodiversity and recommended actions for achieving them. PAs = protected areas.

Component	Status of implementation	Recommended actions
1 Coverage	Medium	Develop globally consistent and comprehensive methods of defining the boundaries of freshwater ecosystems, for accurate measurement of extent covered/not covered by PAs. Ensure that globally at least 17% (ideally more) of freshwater ecosystems are within PAs under the principles in 2 to 5 of this table.
2 Areas of importance	Medium	Implement existing approaches, including KBAs, to identify areas of importance for biodiversity and/or ecosystem services. Integration of these approaches to identify areas where freshwater ecosystems are also optimally located for provision of ecosystem services. Ensure that the freshwater ecosystems included in PAs (see 1 above) include sites of high importance for global biodiversity or ecosystem services provided (see also 5, below).
3 Effective management	Medium	Assess existing strengths and weaknesses of management of freshwater ecosystems in PAs and use this information to (i) achieve better management of existing PA; and (ii) select new protected areas that are not only important for biodiversity or ecosystem services (2 above), but are also expected to be practically manageable. Develop and apply better tools for assessing management effectiveness.
4 Equity	Low	Identify the relevant stakeholders who can represent the full spectrum of equitable management of freshwater resources in PAs. These should include appropriate proxies for ensuring that non-human life needs are represented. Further integration of equitable management as key components of international agreements and Conventions (e.g. Ramsar, UN Watercourses Convention).
5 Ecological representation	Low	Select PAs to include freshwater ecosystems that are important (see 2 above) and manageable (see 3), should be representative of different ecosystem types around the world, in terms of the species present and the geophysical characteristics of the ecosystems.
6 Well connected systems	Low	Develop better spatial tools for assessing longitudinal, lateral, and vertical connectivity of freshwater systems. Identify areas that could be priorities for maintaining connection between PAs. Forward planning to ensure connectivity of freshwater ecosystems that will be modified, in extent and character, by climate change.
7 Other effective area based conservation measures	Low	Define and identify potential OECM sites, and develop guidelines for their management.
8 Integration in the wider landscape	Medium	Freshwater management must become a stronger, required part of decision-making by land managers. Application of place-based management strategies such as those proposed by Abell *et al.* (2007), which equitably support (see section 4) the multiple uses and needs of the catchment.

help to overcome many of the challenges outlined in the section on clause 3. IWRM and basin management planning focus on the use and needs of water, and the management of the resource through demand management, pricing, water conservation measures, and infrastructure. Infrastructure, such as diversion channels, irrigation systems, drainage, flood defences, reservoirs and dams help to maximize the economic benefits from freshwater systems, and underpin economic development, but often to the detriment of freshwater ecosystems and the species within and reliant upon them (Sadoff *et al.*, 2015). Greater integration of protected area benefits with practical water management is needed to maximize the potential for biodiversity protection, ecosystem service provision, and wider freshwater system connectivity at catchment scales.

Measuring and assessing the role of OECMs in meeting Target 11 is also essential (see the section on clause 7). OECMs can be important in facilitating connectivity between protected areas and particularly for flowing waters in places where designating a protected area is not appropriate, or where the efficacy of existing protected areas is compromised. One advantage of OECMs is that they may be more politically straightforward to implement.

Finally, taking into consideration the synergies and trade-offs between Aichi Biodiversity Targets is fundamental for a strategic allocation of resources to achieve them (Marques *et al.*, 2014). Assessments not focused specifically on fresh water have explored how achieving Aichi Target 11 can contribute to Target 12 (Butchart *et al.*, 2012), and the synergies and trade-offs between Targets 11, 12, 5 and 14 (Di Marco *et al.*, 2015).

Beyond 2020

The 20 Aichi Biodiversity Targets as described in the Strategic Plan for Biodiversity are to be met by 2020. Beyond 2020, countries have agreed Sustainable Development Goals (SDGs) which are likely to drive the conservation agenda in the coming decades (UN, 2015). The SDGs specifically refer to fresh water and biodiversity in Goals 6 and 15. Goal 6 is focused on water, and includes the Target 6.6: 'by 2020, protect and restore water-related ecosystems, including mountains, forests, wetlands, rivers, aquifers and lakes.' Target 15.1 aims, by 2020, to 'ensure the conservation, restoration and sustainable use of terrestrial and inland freshwater ecosystems and their services, in particular… wetlands, in line with obligations under international agreements.' Hence, the points made above in relation to the Aichi Biodiversity Targets will have long-term relevance to global policy at least through to 2030, the deadline for achieving the SDGs.

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