A Global Assessment of Inland Wetland Conservation Status

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Wetlands have been extensively modified by human activities worldwide. We provide a global-scale portrait of the threats and protection status of the world's inland wetlands by combining a global map of inundation extent derived from satellite images with data on threats from human influence and on protected areas. Currently, seasonal inland wetlands represent approximately 6% of the world's land surface, and about 89% of these are unprotected (as defined by protected areas IUCN I–VI and Ramsar sites). Wetland protection ranges from 20% in Central and 18% in South America to only 8% in Asia. Particularly high human influence was found in Asia, which contains the largest wetland area of the world. High human influence on wetlands even within protected areas underscores the urgent need for more effective conservation measures. The information provided here is important for wetland conservation planning and reveals that the current paradigm of wetland protection may be inadequate.

Keywords: wetlands, inundation map, human influence, protected areas, conservation planning

Wetlands are highly productive and biodiverse ecosystems (Keddy et al. 2009). They provide many ecosystem services, including water purification, the buffering of runoff and river discharge, the production of food and fiber, and ecotourism (Mitsch and Gosselink 2000, Keddy 2010, Junk et al. 2013). Owing to their high productivity, fertile soils, and importance for provision of water, many of the world's wetlands have historically been occupied and intensively used by humans. Wetlands remain a source of sustenance for local populations, especially in developing countries, and are highly valued by many traditional cultures (Keddy 2010, Maltby and Acreman 2011, Gopal 2013).

In spite of the ecosystem services they provide, wetlands have been lost, degraded, or strongly modified worldwide (box 1). The reported long-term loss of natural wetlands averages between 54% and 57%, reaching up to 90% in some regions of the world (Junk et al. 2013). As was shown by Davidson (2014) in a compilation of 169 reports of historical wetland loss, the extent of inland wetlands declined by 69%–75% in the twentieth century, whereas coastal wetlands declined 62%–63%. The vast decline in both area and ecological condition of wetlands due to human uses and land reclamation is widely recognized and has led to efforts across the world for their protection or restoration (Gardner et al. 2015).

However, measures to ensure wetland protection have not always been effective. The Ramsar Convention on Wetlands (*www.ramsar.org*), the most important international initiative for wetland protection, is a treaty adopted in 1971 with the objective of recognizing the importance of wetlands and promoting their conservation. However, the successful conservation of designated Ramsar sites relies on the investment of local and regional governments as well as international cooperation and can result in variable levels of protection among sites (Gardner et al. 2015). In addition, protected areas are often not designed in a conservation-planning framework (Abell et al. 2007) that incorporates the processes that sustain the functioning of wetlands and the ecosystem services they provide. In particular, hydrological dynamics, ecological processes, and biodiversity should be key features of reserve design and boundaries, but protected areas are often designated with inadequate consideration of the role of upstream (catchment) areas as water and nutrient sources, hydrological connectivity with rivers or other water bodies, wildlife habitat needs and migration corridors, and natural disturbance processes (Saunders et al. 2002, Nel et al. 2007, Pittock et al. 2015, Abell et al. 2016).

Assessing the conservation status of global wetlands is challenging because information on wetland distribution is geographically variable and because existing inventories differ greatly in wetland definitions (broad versus restricted scope), resolution (local versus regional scale), and the accuracy of wetland delineations, making it difficult to compare regions to detect broadscale trends in wetland status (e.g., Mitsch 1994, Finlayson and Van der Valk 1995,

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Box 1. Examples of broad- and local-scale threats to wetlands across the world

Floodplain wetlands have been lost worldwide, such as this tidal floodplain of the Ganges River in Bangladesh that has mostly been converted to rice paddies and aquaculture. However, intact mangrove forest remains in the Sundarbans National Park, where it appears dark green in this Digital Globe satellite image.

River floodplains are often permanently inundated by dams constructed on low-gradient rivers, such as in this example showing the Funil reservoir on the Grande River in the State of Minas Gerais in Brazil, where the trees could not tolerate continuous flooding (photograph: Emma Rosi).

Many inland wetlands have been drained for agriculture, often requiring tile fields and drainage ditches; this photo from Michigan shows the installation of new drainage lines (photograph: Stephen Hamilton).

Invasive plants can strongly impact wetlands, such as this Great Lakes coastal wetland in Michigan that has become dominated by the Common Reed (*Phragmites australis*) (photograph: Todd Marsee).

McComb and Davis 1998, Lehner and Döll 2004, Zedler and Kercher 2005, Junk et al. 2013). There have been systematic inventories of wetland distribution and status in the United States (the US Fish and Wildlife Service National Wetlands Inventory) and Canada (the Canadian Wetland Inventory), and similar but more limited information exists for Australia (the Australian Wetlands Database and the Directory of Important Wetlands in Australia) and some European countries (Annex I of the Habitats Directive [92/43/EC] and the Manual of European Union Habitats–EU-27 [July 2007]). However, most national wetland inventories tend to consist of little more than lists of major wetlands, and the boundaries, if delineated, may not be consistent or accurate, reflecting a significant knowledge gap for wetlands in many of the regions of the world (Finlayson and Van der Valk 1995, Finlayson et al. 1999, Junk et al. 2013, Davidson 2014). Understanding and mapping wetland distribution for broadscale assessments are therefore an important first step toward defining and prioritizing specific conservation needs (Nel et al. 2009, Vörösmarty et al. 2010).

In this study, we provide a global assessment of inland wetland distribution, conservation status, and threats from human pressure. Our specific objectives are to (a) quantify the global distribution of inland wetlands (excluding open-water systems, such as rivers and lakes) across seven geographical units using the best data sources available;

(b) investigate the current protection status of the major wetlands; and (c) estimate human pressure on wetlands, including both unprotected and protected areas. On the basis of this assessment, we provide recommendations to improve planning and management practices for wetland conservation.

Quantifying the global distribution and conservation status of inland wetlands

Wetlands have been defined in varying ways depending on the purpose (e.g., scientific versus regulatory) and the geographic context. Many wetland definitions are present in the literature. Some are more restricted in scope, such as those based on functional characteristics where deepwater habitats (e.g., lakes, reservoirs, rivers, and coastal waters) that are permanently flooded and do not support aquatic vegetation are generally not considered wetlands (Lewis et al. 1995, Mitsch and Gosselink 2000). Other definitions are broader in scope, such as the Ramsar definition that includes all inland waters and the coastal zone up to 6 meters (m) deep as wetlands (Ramsar 2013). In this study, we focus on quantifying seasonally flooded inland wetlands, excluding coastal waters and most deepwater habitats corresponding to rivers, lakes, and reservoirs larger than 300 m in width (as we discuss below).

To quantify the distribution of inland wetlands, we used GIEMS-D15 (Fluet-Chouinard et al. 2015), a downscaled version of the Global Inundation Extent from Multi-Satellites (GIEMS) database (table 1; Prigent et al. 2007). GIEMS is a spatial database summarizing 12 years of satellite observations of surface-water coverage at 0.25-degree resolution, and GIEMS-D15 has been downscaled by Fluet-Chouinard and colleagues (2015) to a resolution of 15 arc-seconds (approximately 500 m at the equator) to show surface-water inundation at three different stages: (1) mean annual minimum, (2) mean annual maximum, and (3) a long-term maximum derived by combining the average 3-year maximum of the GIEMS record with historic wetland extents depicted in the Global Lakes and Wetlands Database (GLWD, Lehner and Döll 2004). GIEMS-D15 provides a comprehensive estimate of seasonal and interannual variation in inundation area, without distinguishing between natural and artificial causes (Fluet-Chouinard et al. 2015). In the present study, to represent the typical water extent over the 12 years of the GIEMS-D15 database, the mean annual minimum and the mean annual maximum surface-water extents were used for the wetland mapping and area estimations. A summary of the data sets used in this study can be found in table 1.

In this effort, we focus on seasonally flooded wetlands, which have been poorly assessed in other freshwater-focused evaluations (e.g., Abell et al. 2016). We removed rivers, lakes, and reservoirs from the wetland area estimates using the Global Width Database for Large Rivers (GWD-LR) of Yamazaki and colleagues (2014; table 1). The GWD-LR database is a global water mask containing rivers, lakes, and reservoirs larger than 300 m in width (for details, see Yamazaki et al. 2014). The temporal variability of river width is not represented in GWD-LR; therefore, after excluding rivers and lakes by masking out the areas of water bodies in GWD-LR, the seasonally inundated areas associated with rivers and lakes remained because they appear in the inundation extents of GIEMS-D15. Wetlands located along the marine coastline, which are not well characterized by GIEMS-D15 because of radiometric interference from the ocean as well as tidally driven variability in inundation extent, were excluded using a 3-kilometer (km) shoreline buffer. The result of these procedures is the inland wetland map presented in figure 1, which provides the basis for estimating wetland protection status and human pressure (see supplemental materials for a higher-resolution figure).

To indicate the protection status of the inland wetlands, we used the World Database on Protected Areas (WDPA),

Table 1. A summary of the main characteristics of the global data sets used in the presented study.

Databases	Main characteristics	Authors
GIEMS	Global Inundation Extent from Multi-Satellites database. GIEMS is a spatial database summarizing 12 years of satellite observations of surface-water coverage at 0.25-degree resolution.	Prigent et al. 2007
GIEMS-D15	GIEMS-D15 is a version of GIEMS downscaled to 15 arc-seconds (approximately 500 m at the equator) supplemented with historic wetland extents depicted in the Global Lakes and Wetlands Database (GLWD, Lehner and Döll 2004). The surface-water extent is shown at three different stages: (1) mean annual minimum extent, (2) mean annual maximum extent, and (3) long-term maximum extent.	Fluet-Chouinard et al. 2015
GWD-LR	Global Width Database for Large Rivers. The GWD-LR database was developed using baseline global water masks containing rivers and lakes larger than 300 m in width. The temporal variability of river width is not represented because only one water mask was used in the calculation of river widths.	Yamazaki et al. 2014
WDPA	The World Database on Protected Areas is the most comprehensive global spatial database on marine and terrestrial protected areas.	IUCN and UNEP–WCMC 2015
Global Human Footprint Map, Last of the Wild Project, version 2	Calculates a Human Influence Index (HII) from nine global data layers covering human population pressure (population density), human land use and infrastructure (built-up areas, night-time lights, and land use or land cover), and human access (coastlines, roads, railroads, and navigable rivers).	WCS and CIESIN 2005

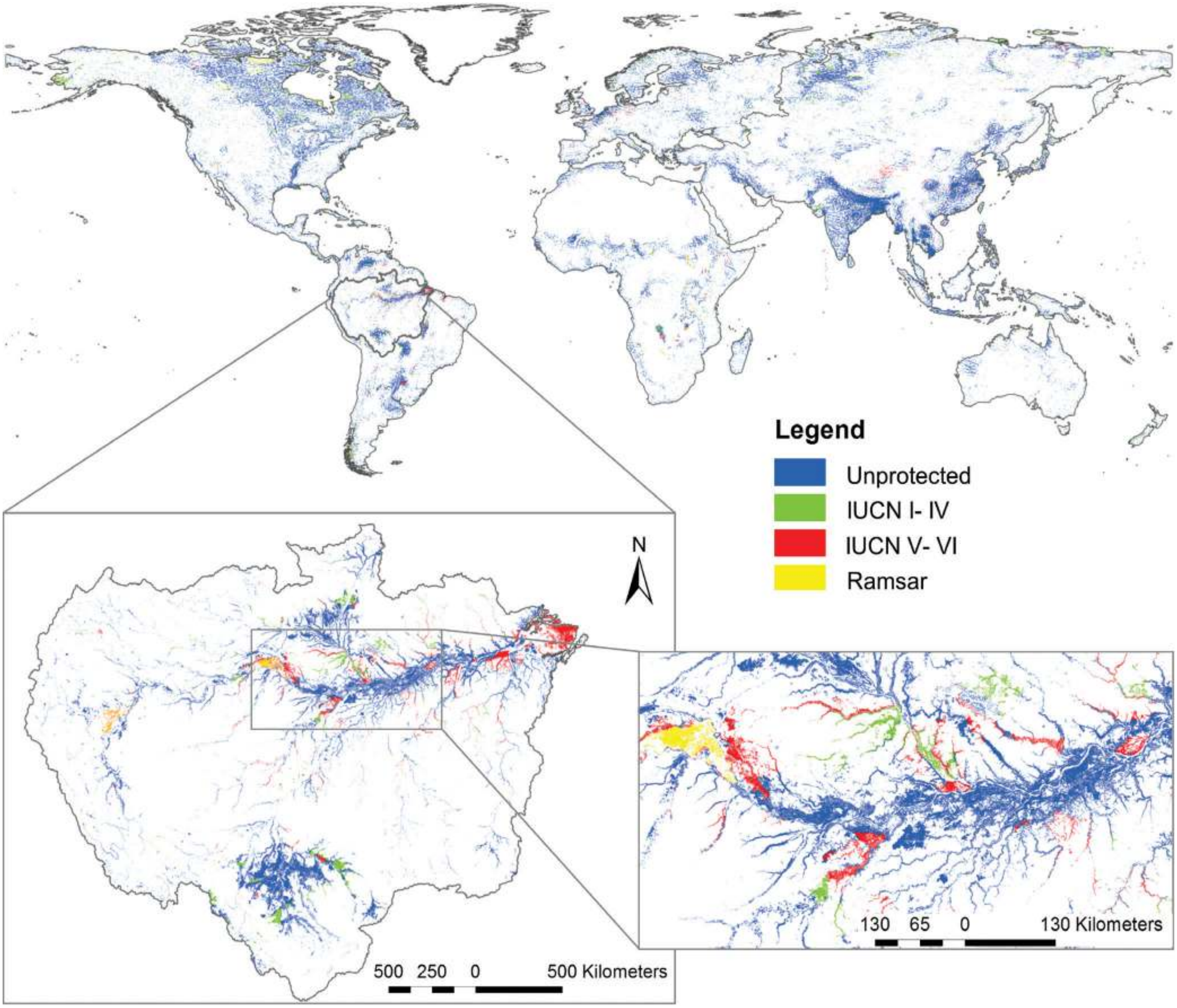

Figure 1. The global extent of inland seasonal wetlands, showing an enlargement of the Amazon River basin and the Central Amazon as examples to show the distribution of unprotected and protected wetlands.

which is the most comprehensive global spatial database on marine and terrestrial protected areas (IUCN and UNEP–WCMC 2015). To evaluate the levels of protection offered by different categories of protected areas, we used only polygons of protected areas assigned to one of the IUCN management categories (I–VI) in the WDPA database. IUCN categories I–IV constitute areas of stricter protection, whereas categories V–VI are areas that allow economic activities. Ramsar sites, despite not being protected under strict IUCN guidelines, were also included in the data analysis because they are specifically designated for wetland management and conservation. We did not include other protected areas of the WDPA database that lacked information on management objectives (i.e., those not labeled IUCN I–VI or Ramsar). The total land area of categories I–VI and Ramsar sites of 14.9 million square kilometers (km^2) corresponds to 67% of all designated terrestrial sites in the WDPA database. Overlaying the polygons of the protected areas on the inland wetland map, we quantified the area of wetland within protected areas of IUCN categories and Ramsar sites.

Overlaying the protected areas onto the global inland wetland map reveals that 6.1% of the Earth's land surface (excluding Antarctica) is covered by seasonal inland wetlands, but 88.7% of these are unprotected as defined by IUCN categories I–VI and Ramsar sites. Among the 11.3% of protected wetlands, 9.7% are within the IUCN categories I–VI, including 7.1% in the more strictly protected categories I–IV and 2.7% in the categories V–VI. The remaining 1.5% are designated as Ramsar sites (table 2). Terrestrial protected areas often include inland waters such as rivers, lakes, and wetlands. Recent estimates of terrestrial protected area coverage range from 15.4% (Juffe-Bignoli et al. 2014) to

Table 2. A summary of the total wetland area, the total wetland in protected areas (PA), and the percentage of wetland protection in each geographical unit.

Geographical units	Total wetland area (km²x10⁶)	Total wetland in PA (km²x10⁶)	Total percentage of wetland protection	Percentage of wetland protection per category of PA		
				IUCN I–IV	IUCN V–VI	Ramsar
South America	0.89	0.16	17.82	7.17	9.09	1.56
North America	2.46	0.30	12.24	9.30	0.94	2.00
Central America	0.04	0.01	20.30	14.87	2.82	2.62
Africa	0.74	0.09	12.88	6.18	3.35	3.35
Europe	0.75	0.11	14.93	7.87	4.63	2.42
Asia	4.11	0.33	8.01	5.03	2.26	0.72
Oceania	0.17	0.03	15.49	10.63	3.58	1.28
Total	9.16	1.03	11.25	7.05	2.69	1.52

14.7% (Harrison et al. 2016) of the Earth's land surface, considering different subsets of protected areas in the WDPA database. The terrestrial protected-area coverage evaluated in this study corresponds to an area of 14.9 million km^2 (approximately 10% of the land surface), within which the inland wetlands amount to 1.0 million km^2 (11.3% of the total wetland area; table 2).

Our estimate of globally protected wetland area is lower than a previous estimate of 20.7% from Juffe-Bignoli and colleagues (2014). However, those authors used a broader definition of wetlands, including rivers, lakes, and reservoirs, and used the Global Lakes and Wetlands Database (GLWD; Lehner and Döll 2004). They also considered a larger coverage of protected areas corresponding to a total area of 20.6 million km^2, using all records in the WDPA database, including protected areas without an assigned IUCN category.

The IUCN categories I–IV account for most of the protected wetland area (table 2). However, inadequate consideration of the ecological functioning of freshwater ecosystems when designing terrestrially focused protected areas could compromise the effectiveness of conservation efforts for freshwater ecosystems in general and wetlands in particular (Abell et al. 2007, Pittock et al. 2015). Protected areas originally designed to protect terrestrial ecosystems may not adequately conserve aquatic ecosystems when they do not consider the importance of hydrological and biotic connectivity in the upstream, downstream, lateral, and temporal dimensions (Pringle 2001, Pittock et al. 2015). In the past decade, many authors highlighted the need for a paradigm shift in the design of freshwater protected areas, calling for integrated river-basin management plans, including spatially explicit conservation visions, targets, and strategies to ensure the conservation of freshwater ecosystems and the services they provide (Abell et al. 2007, 2016, Nel et al. 2007, Linke et al. 2011).

The fraction of wetland protection varies greatly among geographical regions (table 2). Central and South America present the largest percentages of wetland protection, at 20.3% and 17.8%, respectively. The coverage by individual categories of protection, however, tells a different story: In South America, 7.2% of the wetlands are protected in IUCN categories I–IV, 9.1% are protected in the lower-tier IUCN categories V–VI, and 1.6% are Ramsar sites. In Central America, 14.9% of the wetlands are within the strictly protected IUCN categories I–IV, only 2.8% are in IUCN categories V–VI, and 2.6% are Ramsar sites. In Asia, which is the geographical region with the lowest fractional wetland protection, only 5.0% are in the IUCN categories I–IV, 2.3% are in IUCN categories V–VI, and 0.7% are Ramsar sites. The percentage of protected wetlands in Asia is potentially an underestimate because of the presence of rice paddies, which are not distinguished from natural wetlands in GIEMS-D15. Harrison and colleagues (2016) reported similar estimates of the contribution of protected areas to global water provision for some of those regions. Whereas South America derives one of the largest volumes of freshwater provision from protected areas, Asia derives one of the smallest volumes.

Asia and North America contain the largest extents of wetlands: 4.1 million km^2 and 2.5 million km^2, respectively (table 2). However, the level of protection in these regions differs greatly, and much of the wetland area is in boreal and Arctic latitudes. In North America, both Canada (Mitsch and Hernandez 2013) and the Midwest in the United States (Dahl 1990) reported wetland losses of approximately 80%, mostly by drainage for agricultural production, in the past 100 years. Following this long-term history of wetland drainage and degradation in North America, matters have improved since the 1980s through restoration and mitigation efforts (Dahl 2006, Davidson 2014). Although wetland losses may have been at least partially offset by restoration and wetland creation in rural and suburban areas since the mid-1980s in the United States (Dahl 2006, Dahl and Stedman 2013), scientific debate remains as to how well artificial wetlands replace natural ones (Zedler and Kercher 2005).

Asia is home to more than one-third of the Earth's human population, and wetlands are suffering the consequences of intensifying pressures to meet human demands (An et al. 2007, Gopal 2013). More than 75% of the world's rice-paddy fields are situated within tropical Asia, many representing converted wetlands (e.g., box 1). Deforestation, drainage, and reclamation of wetlands (including peat swamps) for conversion to agriculture, fish culture, or other uses, as well as dam construction on rivers with floodplains, have been increasing threats to Asian wetlands in the last 60 years. Current loss rates for Asian wetlands are estimated at about 5000 km^2 per year (Zedler and Kercher 2005, Gopal 2013). The mounting pressures on wetlands in Asia and the small fraction of protected wetlands call for urgent action.

Threats to inland wetlands

There is a strong association between wetland distribution in the landscape and human occupation, with the most significant threats to wetlands being associated with direct or indirect human use of these areas (Gibbs 2011). To quantify the local human pressure in wetlands, we used the Global Human Footprint Map, Last of the Wild Project, version 2 (table 1; WCS and CIESIN 2005), which calculates a Human Influence Index (HII) from nine global data layers covering human population pressure (population density), human land use and infrastructure (built-up areas, nighttime lights, and land use or land cover), and human access (coastlines, roads, railroads, and navigable rivers). This index, mapped at a 1-km resolution, varies from 0 to 100 representing the intensity of the human pressure normalized by biome and realm (Sanderson et al. 2002). To quantify the human pressure on wetlands, we obtained the distribution of values of HII for all wetland pixels in our global wetland map (figure 1), and we summarized HII by protection status and geographical region.

Human pressure in wetlands varies widely among geographical units (figure 2), with the lowest average HII values in Oceania (12.6 ± 13.1) and North America (12.3 ± 19.5) and the highest values in the wetlands of Europe (34.0 ± 21.9), Central America (34.7 ± 13.7), and Asia (27.1 ± 18.7). The low human influence for North America can be attributed to the large area of boreal and Arctic wetlands in Canada and Alaska. The mean HII of the wetlands in the lower 48 states of the United States is even higher than in Europe and Central America (39.1 ± 21.8). Despite having 20.3% of its wetlands protected, Central America suffers from high rates of deforestation and conversion of forest to agriculture. In addition, population and development pressures formerly restricted to upland areas are expanding rapidly into wetlands, resulting in wetland losses at rates comparable to tropical forest degradation (Ellison 2004). In Europe, historical wetland degradation caused by human occupation and use within and surrounding wetlands largely accounts for the high levels of human pressure (Čížková et al. 2013). Vörösmarty and colleagues (2010) found similar patterns in regions of intensive agriculture and dense settlement, such as much of the United States, all of Europe, and large portions of Asia, when analyzing threats to freshwaters for human water security and biodiversity.

Comparing wetlands within different categories of protected areas, IUCN categories I–IV provide the most protection from human pressure, whereas categories V–VI provide the least (figure 3). However, this difference may also reflect preexisting differences in occupation and use that influenced the selection of the protection categories. Ramsar sites have an intermediate level of human pressure compared with the two IUCN category groups (I–IV and V–VI). The exception is Central America, where the Ramsar sites are more affected by human pressure than are the wetlands in IUCN categories V–VI. In contrast, Ramsar sites in Oceania are generally less affected by human pressure than are the wetlands in IUCN categories I–IV. In some instances, high levels of human pressure were found within protected areas, such as in IUCN categories V–VI in Europe (figure 3).

One-fifth of the global runoff is generated within protected areas, and the presence of threats in protected areas and their surroundings affects the quality of water provision from these regions, as has been demonstrated by Harrison and colleagues (2016). They found that the majority of the Earth's protected areas are not highly threatened, although they reported the occurrence of significant threats in a large percentage of the protected areas of Asia, Europe, and North America. However, Harrison and colleagues (2016) did not evaluate the levels of protection provided by the different categories of protected areas.

Variable levels of human pressure in Ramsar sites, such as the contrasting results we report here for Central America and Oceania, could be explained by the fact that variable governance of Ramsar sites offers different levels of protection in different regions, and a range of activities can take place in these areas (Gardner et al. 2015). Australia was the first country to sign the Ramsar Convention in 1971, and it developed a national wetland policy in 1997 to manage wetlands on the basis of the Ramsar principles of "wise use" that target the sustainable use of resources. In Central America, all countries became signatories to the Ramsar Convention in 1990, but integrated planning for the management and conservation of wetlands in the region only began in 2002 (Ellison 2004). Ramsar sites often lack practical management plans or have insufficient ones, leading to ineffective management, which can result in the deterioration of the ecosystem services and ecological condition of wetland sites (Gardner et al. 2015). This is also an issue for many protected areas in general and can severely affect freshwaters in particular (Thieme et al. 2012, 2016).

In summary, our analyses demonstrate that human pressure on wetlands is variable according to the geographical region and that some protected wetlands are subject to significant human pressure. Similar conclusions have been reached in previous studies of wetlands (Pittock et al. 2015)

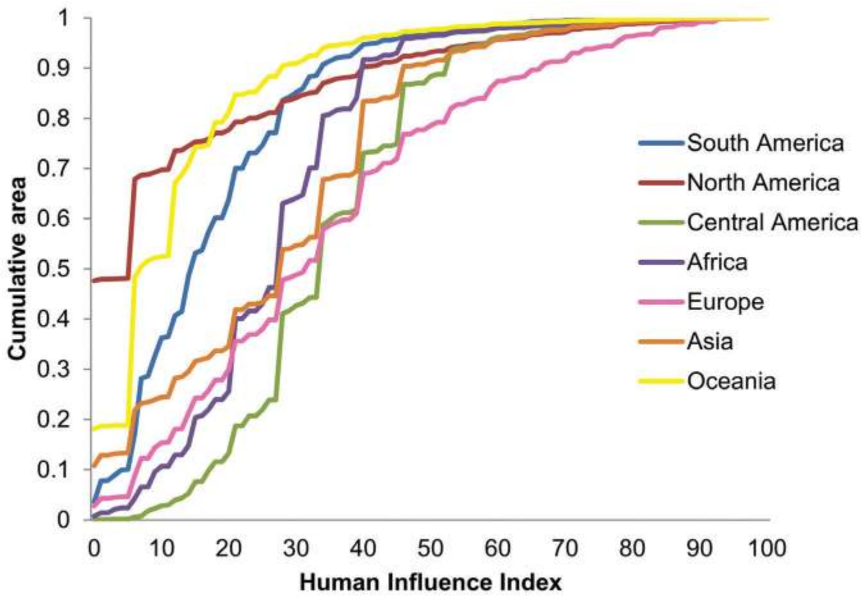

Figure 2. Human pressure in wetlands per geographical unit, as is indicated by the cumulative area under a given Human Influence Index (0, lowest; 100, highest human influence).

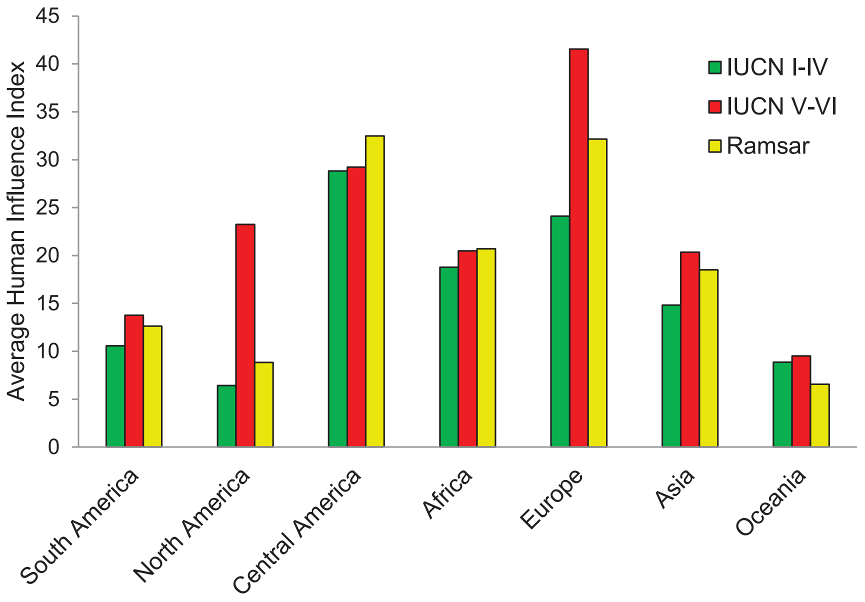

Figure 3. Human pressure in wetlands within different categories of protected areas, as is indicated by the average Human Influence Index (0, lowest; 100, highest human influence).

and other freshwater ecosystems in protected areas (Abell et al. 2016). It is concerning that only a small fraction of global seasonal wetlands is covered by protected areas (11.3% overall; table 2) and much less under the stricter IUCN categories I–IV. Adding to that, high levels of human influence in some of the protected wetland areas (see figure 3) indicate that the local ecological condition of protected wetlands may also be compromised. Although beyond the scope of this study, freshwater ecosystems are also threatened in protected areas because of the propagation of impacts by hydrological and biotic exchanges from upstream or downstream areas (Thieme et al. 2012, 2016, Abell et al. 2016). Impacts from outside protected areas such as altered flow regimes, deteriorated water quality by pollution, sedimentation, and invasive species can seriously affect the ecological condition of freshwater systems (see box 1 for examples). Consequently, these impacts affect the provision of essential ecosystem services, such as water provision for people (Harrison et al. 2016). In that context, it is important to consider that our assessment of human influence using the HII only incorporates local threats (i.e., those generated within the wetlands) and does not consider the potential negative influences of human activity propagating into wetlands from outside. Given the potential of exogenous drivers of impact, wetlands might be more severely threatened than we report here (e.g., as has been discussed by Pittock et al. 2015, Harrison et al. 2016, Thieme et al. 2016). A more comprehensive estimate of impacts on wetlands would require site-specific studies that include consideration of exogenous pressures that may propagate to wetlands. Wetlands smaller than 0.25 km^2, not represented herein because of the limitations of global data sets and their spatial resolution, might suffer from similar pressures and also need to be considered in local and regional conservation planning.

Moving forward: Suggested improvements in wetland conservation planning and practice

Protected areas should be designed to maximize the effectiveness of the reserve systems to protect biodiversity and ecosystem services. Fundamental principles have been proposed to guide effective design networks of protected areas (e.g., representativeness, Austin and Margules 1986; persistence, Soulé 1987; the CARE principles of cost, adequacy, representation, and efficiency, Possingham et al. 2006; and protected-area governance and management, Worboys et al. 2015). In that context, the growing field of systematic conservation planning seeks to identify configurations of complementary areas that achieve explicit and generally quantitative conservation objectives (Pressey et al. 2007) while accounting for cost-effectiveness and spatial constraints (Watson et al. 2011).

Despite the existence of guidelines and tools for reserve design, most of the existing reserve systems throughout the world were declared without following these principles, and as a result, they may inadequately represent ecosystems and biodiversity (Rodrigues et al. 2004, Juffe-Bignoli et al. 2014). Reserves have usually been established in places that had already proven unsuitable for commercial activities and often were designed using *ad hoc* methods (Pressey et al. 2002). The case for freshwater ecosystems is even further complicated because of the fundamental importance of hydrological connectivity, which until recently was mostly neglected in reserve design, with rivers often being used as boundary limits for terrestrial reserves (Nel et al. 2007) and wetlands often completely ignored. For these reasons, freshwater ecosystems are more imperiled than their marine or terrestrial counterparts (Saunders et al. 2002, Strayer and Dudgeon 2010, Pittock et al. 2015).

A few studies have applied the principles of systematic conservation planning to address wetland conservation needs. Lourival and colleagues (2009) evaluated the representativeness, efficiency, and complementarity of four preexisting conservation scenarios proposed by the Brazilian government to protect the Pantanal wetlands and found that none of the proposed scenarios would protect all conservation targets, presenting suggestions for better planning to protect those wetlands. Ausseil and colleagues (2011) mapped the historical and remaining wetlands in New Zealand and applied the principles of systematic conservation planning to prioritize wetland conservation efforts. They considered wetland condition, representativeness, and complementarity in different biogeographic contexts and proposed a ranking reflecting the potential to protect different wetland classes. Schleupner and Schneider (2013) proposed a model to allocate wetland restoration efforts in Europe considering spatially explicit wetland distribution data and environmental and economic constraints. These new approaches represent important steps toward the integration of freshwater and terrestrial conservation planning.

However, the studies above have often ignored important factors that affect wetland persistence in the landscape, such as (a) the role of connectivity between wetlands and with other water bodies in exchanging water, nutrients, and organisms; (b) spatiotemporal variation in water depth and extent that affects wetland functions; and (c) the propagation of impacts from upstream to downstream in catchments and in wetlands connected to rivers (figure 4). Such factors can increasingly be addressed in wetland conservation planning as the appropriate data and methods are made available.

Innovations in the field of freshwater conservation planning can be adopted to improve conservation planning for wetlands, especially regarding the maintenance of natural patterns of hydrologic and biotic connectivity between waterbodies in the landscape. New methods to deal with the longitudinal connectivity of rivers in reserve selection have been developed (Moilanen et al. 2008, Hermoso et al. 2011), providing a toolbox to facilitate ecologically meaningful spatial conservation prioritization for river networks. Furthermore, Hermoso and colleagues (2012a) proposed a new approach defining and applying multiple connectivity rules for freshwaters to enhance the lateral and longitudinal biotic exchanges within priority areas. Spatiotemporal connectivity was incorporated in freshwater conservation prioritization, including consideration of temporal connectivity and the refugial role of freshwater bodies for aquatic life during dry periods (Hermoso et al. 2012b, 2013). Likewise, Rivers-Moore and colleagues (2016) suggested a connectivity index for rivers that can be used to prioritize important freshwater areas that would be better protected by maintaining or enhancing connectivity between waterbodies in catchments.

Similar approaches could be used to enhance wetland connectivity in systematic conservation planning for freshwaters. For example, wetland protection could be enhanced by the inclusion of longitudinal and lateral components in reserve selection to account for the influence of river networks and the impacts of water flowing into wetlands from their catchments. Similarly, the consideration of spatiotemporal dynamics of wetlands in reserve selection could help preserve ecosystem services related to the hydrological regime of different wetlands. Such measures would provide a sounder ecological background to allocate wetland conservation efforts and enhance catchment-integrated management.

Dynamic frameworks are needed in systematic conservation planning to account for the temporally variable connectivity of natural wetland mosaics in the landscape (figure 4). For example, wetlands seasonally connected to rivers, such as floodplains, require planning for the maintenance of lateral as well as longitudinal connectivity that may occur seasonally or episodically, and their ecological condition may be directly associated with the river-system health. Other types of wetlands may be hydrologically isolated and dependent on groundwater discharge or rainfall regimes. Conservation of groundwater-dependent wetland types may require the preservation of their surroundings to maintain groundwater flow. These wetlands may also be exposed to different types of impacts originating from their terrestrial catchment (figure 4). Where isolated wetlands are subject to occasional desiccation in dry years, preserving connectivity with refugial habitats that remain wet within protected areas can be important to maintain aquatic fauna (Hamilton et al. 2005, Davis et al. 2013). Therefore, understanding how connectivity governs wetland structure and functioning in the landscape is a fundamental step toward building more effective schemes to include wetlands in systematic conservation planning.

Furthermore, conservation planning must consider both ongoing natural and human disturbances to better deal with

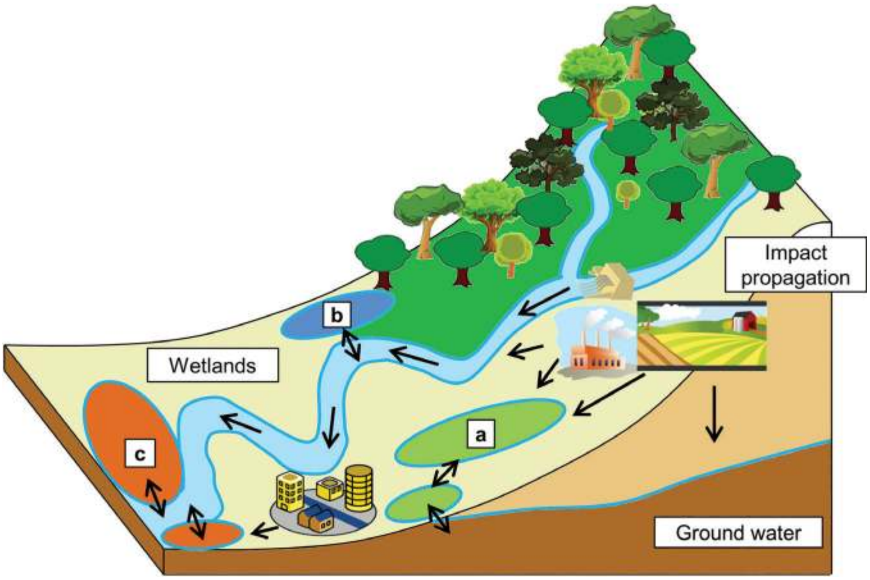

Figure 4. A schematic representation of wetland hydrological connectivity in the riverine landscape and impact propagation from human activities in the upper catchment: (a) wetlands isolated from the river channel, (b) wetlands temporarily connected to the river channel, and (c) wetlands permanently connected to the river channel.

the reality that human activities are vastly altering the planet; consequently, biodiversity is also changing in time and space (Pressey et al. 2007, Game et al. 2013, Magurran 2016). This is especially important in the case of wetlands that draw from a species and resource pool in a parent river or where refugial habitats are key to sustaining local populations. In such cases, connectivity is key to allow species migrations, range shifts (e.g., in response to climate change), and long-term evolutionary processes (Davis et al. 2013). Indeed, a more integrated view of conservation must be taken for effective long-term wetland protection, considering all parts of the freshwater–terrestrial landscape and how processes interact across different timescales (Bornette et al. 1998, Abell et al. 2016).

Finally, measures to reduce the current and increasing human pressures on wetlands are essential, especially in areas where much of the natural wetlands have already been lost or are strongly degraded. The guiding principles of conservation planning can also be applied to prioritize wetland restoration in those areas (e.g., Schleupner and Schneider 2013). Progress in wetland restoration ecology has led to better practices, with consideration of empirical evidence at multiple spatiotemporal scales and different restoration contexts according to the landscape characteristics (Zedler 2000). In many regions of the world, wetland restoration has been demonstrated to improve ecosystem services and benefit local communities. For example, Schindler and colleagues (2014) showed that the restoration and rehabilitation of lowland river floodplains in Europe consistently improved the multifunctionality of these landscapes, enhancing several ecosystem services. Thompson and Balasinorwala (2010) showed the benefits of a restoration program in the Hail Haor wetlands of Bangladesh that improved the economic benefits of fisheries and ecotourism because of the return of resident bird species. Similarly, Almack (2010) showed the multiple benefits of a wetland restoration project to avoid and mitigate floods in the Napa River Basin; the restoration not only reduced the risk of floods but also increased property values and tourism and improved the water quality and wildlife habitat in the region. Such restoration efforts are expected to continue to expand worldwide.

Although there is consensus on the importance of restoration projects, they have largely ignored the impacts of pressures operating beyond the local scale and have focused mainly on the quantification and management of single pressures (Lorenz and Feld 2013, Verdonschot et al. 2013). As we have previously discussed, aquatic ecosystems are affected not only by local pressures but also by processes operating at broader spatial scales. Especially in the case of wetlands, incorporation of actions such as restoring connectivity by providing environmental watering, creating riparian buffers, and reducing point-source pollution locally and upstream can help to increase wetland area and recover important ecological functions in the landscape (e.g., Pfadenhauer and Grootjans 1999, Cui et al. 2009, Berney and Hosking 2016). Restoration of degraded wetlands together with the conservation of wetlands that remain relatively undisturbed can help enhance the conservation of biodiversity and the ecosystem services that wetlands provide.

Conclusions

This study provides a consistent global portrait of inland wetland distribution, conservation status, and human pressure. We show that many wetlands are subject to human pressures, even at times within protected areas, and question whether terrestrial protected areas are always adequate to protect wetland ecosystems. Considering the rapid increase in human population and pressures on global wetlands, urgent action is needed to develop better frameworks for wetland conservation planning. Identifying the specific conservation needs of the different wetland types considering their variation in space and time, as well as their functions and landscape context, will help support the development of more effective conservation plans. This study provides key information to foster discussion toward improving the current paradigm of wetland conservation—a historically overlooked topic.

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Supplemental material

Supplementary data are available at *BIOSCI* online.

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