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Tracking global change in ecosystem area: The Wetland Extent Trends index

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ABSTRACT

We present a method for estimating broad trends in ecosystem area based on incomplete and heterogeneous data, developing a proof-of-concept for the first indicator of change in area of natural wetland, the Wetland Extent Trends (WET) index. We use a variation of the Living Planet Index method, which is used for measuring global trends in wild vertebrate species abundance. The analysis is based on a database containing 1100 wetland extent time-series records and the method identifies and addresses ecological and biogeographic biases in the dataset. Globally, the natural WET index, excluding human-made wetlands, declined by about 30% on average between 1970 and 2008. Declines varied between regions from about 50% in Europe to about 17% in Oceania over the same period. The WET index fills an important gap in the ecosystem coverage of global biodiversity indicators and can track changes related to a number of current international policy objectives. The same method could be applied to other datasets to create indicators for other ecosystems with incomplete global data.

1. Introduction

In order to monitor progress towards policy goals and targets, decision makers require indicators that provide timely, relevant information on biodiversity change (Jones et al., 2011; Pereira et al., 2013). Although global biodiversity assessments are incorporating an increasing number and breadth of indicators (Butchart et al., 2010; Tittensor et al., 2014), the evidence base remains uneven and there are considerable gaps in indicator coverage (Chenery et al., 2015; Tittensor et al., 2014; Walpole et al., 2009).

One of the key gaps to fill relates to the state of natural ecosystems either in terms of their area or condition (Chenery et al., 2015; Jones et al., 2011). Whilst species-level indicators are relatively well-developed (Walpole et al., 2009), global indicators at an ecosystem-level are limited to forest cover (FAO, 2010; Hansen et al., 2013), which are used in a number of multilateral processes including the Ramsar Convention on Wetlands of International Importance, the Convention on Biological Diversity (CBD) and the Millennium Development Goals (UN, 2014a). Despite this gap, international targets relating to ecosystems have been agreed by governments, such as the Aichi 2020 Biodiversity Targets to halve the rate of loss of all natural habitats (Target 5) and to safeguard and restore ecosystems that provide essential services and contribute to health, livelihoods and wellbeing (Target 14) (CBD, 2010).

One of the natural ecosystems for which decision makers currently lack indicators for is wetland (here defined in accordance with Ramsar Convention, 1971). Wetland ecosystems are not only rich in biodiversity (Strayer and Dudgeon, 2010) but also particularly valuable in terms of the services they provide to people including water security, hydrological regulation, erosion control and in support of a range of production sectors (Costanza et al., 2014; de Groot et al., 2012; Russi et al., 2013). Wetlands also support the functioning of other ecosystems and the services they deliver, especially water flows. Given the significance of wetlands for water security, food security and human health, indicators of wetland change are likely to be of value in environment and development policy contexts including the emerging UN Sustainable Development Goals (Griggs et al., 2013; UN, 2014b).

In a recent global review, Davidson (2014) found that as much as 87% of wetland area may have been lost since 1700 CE, while 20th century losses alone have been 64 to 71% of the area present in 1900. Wetlands are threatened by pollution, fragmentation and transformation, resulting in significant degradation and loss of extent (MA, 2005a; Moser et al., 1996; van Asselen et al., 2013; Vörösmarty et al., 2010). The

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previous recognition of these threats and the value of conservation and wise use of wetlands led to the establishment of the Ramsar Convention on Wetlands of International Importance in 1971, which contains specific policy commitments to maintain and restore wetlands and the ecosystem services they provide (Ramsar Convention Secretariat, 2012a). In 2005 the Ramsar Convention endorsed the development of an initial set of eight key indicators of the effectiveness of the implementation of the Convention (Ramsar Convention, 2005). The endorsement included a desire for an indicator on the 'status and trends in wetland ecosystem extent' in order to measure progress towards the Convention's objectives.

In this study, we use available data to construct an initial global index of change in natural wetland extent as a proof-of-concept for a method to create ecosystem area indicators based on the meta-analysis approach of the Living Planet Index (LPI). The LPI collates species population trend data from published peer-reviewed and grey literature and determines the average rate of change in abundance across taxa over time (Collen et al., 2009; Loh et al., 2005). The LPI method is potentially applicable to any data set that includes time-series for a large number of entities, provided that they all measure the same type of thing such as population, money or physical size, even if the data are at different spatial scales and from different time periods. For example the Index of Linguistic Diversity applies the method to numbers of speakers of languages (Harmon and Loh, 2010; Loh and Harmon, 2014).

We explore potential biases in the underlying wetland area data and in the results derived from the use of the LPI method. Finally, we consider how improvements to the initial wetland indicator, the Wetland Extent Trends (WET) index, would enhance its robustness and policy value for reporting both to the Ramsar Convention and also, in relation to Ramsar's lead implementation role on wetlands for the CBD, on Aichi Target 5.

2. Materials and methods

2.1. Approach

The main challenge faced when constructing an indicator of trends in global wetland cover is the inconsistent and uneven availability of time-series data (as is the case for other components of global biodiversity). The wetlands for which data exist are unevenly distributed around the world, reflecting the distribution of research effort rather than the distribution of wetlands, and the periods covered by the available time-series are inconsistent. To address this challenge we used the method employed by the LPI, which aggregates population trends across multiple species where the data are also inconsistent and uneven. The LPI compensates for unevenness by giving equal weight to each species regardless of the number of population time-series available for each species, and for time-series inconsistency by chaining together average annual trends across time-series in each year covered by the index, regardless of the number of time-series available in each year. This still leaves the issue that there are more data available for species from well-studied regions, biomes or taxa, and the LPI compensates by the further use of weightings applied to regions, biomes or taxa when aggregating to the global index.

The difficulty when applying the LPI method to ecosystem area is that there is no equivalent to a species to act as the basic unit of aggregation when constructing the index. One solution would be to treat each individual wetland time-series as the basic unit, giving equal weight to each time-series within a region (e.g., Africa or Europe) and then building a global index from the regional components. However, there remains a great deal of geographic unevenness in the data coverage and therefore bias within regions, as well as between regions. Similarly, there is ecological unevenness because some wetland-types such as peatlands (Joosten, n.d.) and mangroves (FAO, 2007, 2014) have better data coverage than others.

To address these biases in data availability we created an artificial equivalent to a 'species' of wetland for the purpose of constructing the WET index. We divided the world into sub-regions with broadly similar biogeographic characteristics, and we classified all wetland ecosystems into broad wetland classes, based on the Ramsar wetland categories (see Table A1 in the online Appendix). This was done separately for both terrestrial wetlands and marine/coastal wetlands, thereby creating two matrices of sub-region by wetland class, into which we were able to sort all the available time-series data (see Table A2 in the online Appendix). Each cell of the terrestrial and marine matrices became the equivalent of a wetland 'species', and was used as the basic unit in constructing the WET index. Where multiple time-series existed within a single cell, these were averaged into a single trend for that 'species'. This is justified biogeographically and ecologically on the grounds that wetlands of the same class in the same sub-region are likely to contain similar communities of species facing similar threats. That single trend then represented all wetlands of the same class within one sub-region in the WET index.

The wetland 'species' cell trends were first used to generate regional WET indices which were then aggregated to generate the global WET index. This was done separately for inland wetlands, marine/coastal wetlands and for 'natural' wetlands, which combined inland and marine/coastal wetlands. Human-made wetlands were excluded from this analysis. As with the LPI method, the construction of the WET index consists of a number of stages including collecting of time series data, codification and database entry, aggregation into sub-indices, and further aggregation to create sub-global (ecologically and regionally specific) and global indices. These stages are described below.

2.2. Collection of time-series data

Time-series data on wetland extent were collected from a literature search conducted in English using SciVerse's Scopus online peer-reviewed bibliographic database as well as from non-governmental research institutes and directly from relevant experts. The database titles, abstracts and keywords were queried using a search string that consisted of synonyms for "area" combined with synonyms for "change" and different wetland type terms. For details of the full search see Table A3 in the online Appendix.

In order to capture as much of the wetland literature as possible, time-series were included in the analysis if the data met three basic criteria:

1. data were given in units of area;
2. data were available for at least two points in time; and
3. data points were comparable across time such that changes in area over time could be expressed as a ratio.

In total, 1100 time-series from 170 source references were used in the wetland extent database, the full list of which can be found in Reference list A1 in the online Appendix.

2.3. Preliminary processing of time-series data

Wetland area time-series data consisting of an area in a specific year were entered into a database along with the following metadata:

- Ramsar region (e.g. Europe): regional allocation followed the Ramsar Convention Secretariat's classification (2012b) (see Table A4 in the online Appendix);
- Sub-region (e.g. Western Mediterranean);
- Country (e.g. France): country allocation followed the Ramsar Convention Secretariat's classification (2012b);
- Locality for the wetland (e.g. Camargue);
- Ramsar wetland type, either marine/coastal, inland or human-made: allocation followed the Ramsar Convention Secretariat's classification (2014);

- Wetland class (e.g. intertidal wetland): allocation to one of 19 classes (see Table) based on the Ramsar Convention Secretariat's classification(2014) (see Table A1 for comparison); and
- Source reference.

Finally, each unique time-series record was given an ID based on its locality, wetland class and source reference.

Wetland sites consist of multiple wetland classes; coastal wetlands can be a mixture of inland and marine/coastal wetland classes and few wetlands are entirely unaltered by humans. Wetland class and type allocation was made using the information provided in the source reference. The natural wetland types, marine/coastal and inland, are defined as ecosystem types that have not been wholly altered to a non-wetland land-cover type or to a human-made wetland class (see Table 1). It was assumed that the extent data from the source references had been disaggregated to individual wetland classes and wetland types. Where this was not the case, a 'mixed' wetland class or type was assigned.

There was variation in how the time-series were distributed globally amongst the different wetland classes within each of the three wetland types (see Fig. 1). For the distribution of time-series across the wetland classes and sub-regions see Table A2.

2.4. Assigning data to the matrices

Each time-series record was assigned to one of 126 sub-regions in the database, 57 terrestrial (human-made and inland) and 69 marine. Secondly, records were allocated to one of 19 wetland classes (see Table 1), such as permanent shallow marine waters, marshes on peat soils or agricultural wetlands. There are six marine, six inland and four human-made wetland classes, which are a simplification of the 42 Ramsar wetland classification categories. Three other classes of wetland – geothermal, mixed wetlands and unclassified – were included as additional inland wetland classes, making nine inland classes in total. There were therefore 1155 possible combinations of sub-region and wetland class (57 × 13 terrestrial plus 69 × 6 marine) in the matrices (see Table A2).

Table 1
The wetland types and classes used in the analysis.

WET index classification
Marine/coastal
Permanent shallow marine waters
Coastal shores
Estuarine waters
Intertidal wetlands
Lagoons
Mixed marine/coastal wetland
Inland
Flowing water
Lakes, pools & marshes
Shrub or tree dominated wetlands
Marshes on peat soils
Alpine & tundra wetlands
Mixed inland wetland
Human-made
Aquaculture & salt exploitation
Agricultural wetland
Water storage areas
Mixed human-made wetland
Other wetland
Geothermal & subterranean
Unclassified wetland¶
Mixed wetland type‡

¶ Class consists of wetlands from sources with insufficient description to be classified more specifically.
‡ Class consists of wetlands that could not be separated between the three wetland types: marine/coastal, inland or human-made.

A total of 322 cells in the matrices contained at least one time-series record: 121 marine, 139 inland and 61 human-made. Of these 322 sub-region x class combinations, 138 contained only a single time-series record. Where more than one time-series record existed for a single cell, these were aggregated to give a single trend line, or sub-index, using the method described in 2.5 (this is the equivalent of averaging several population trends for a single species in the LPI) and treated as a single time-series in the next step of the analysis.

2.5. Calculation of sub-indices and regional indices

The calculation of the sub-regional x class sub-indices and the regional indices followed the LPI chain method described in Loh et al. (2005) and Collen et al. (2009). The time period of 1970 to 2008 was chosen because the availability of time-series in the database declined sharply either side of those dates. Where wetland time-series lacked values for each year, we interpolated missing annual values from given values by assuming a constant annual rate of change in area:

$$A_y = A_p \cdot (A_s/A_p)^{\left[\frac{y-p}{s-p}\right]} \tag{1}$$

where A is the area of wetland, y is the year for which the value is interpolated, p is the preceding year with a given value, and s is the subsequent year with a given value. No derived values were extrapolated beyond the first and last values of a time-series.

For each wetland we took the ratio between its area in one year and the preceding year and calculated the geometric mean of the ratios (Δ_y):

$$\Delta_y = \sqrt[N]{\prod_{i=1}^{N} \frac{A_{iy}}{A_{iy-1}}} \tag{2}$$

where A_{iy} is the area of wetland i in year y, and N is the number of wetlands with values for area in both years y and y − 1. To avoid dividing by zero, the area of each wetland in each year was increased by one hectare. Adding one hectare represents an increase of less than 1% for most wetlands in the database. Very large positive or negative annual area change ratios in a single wetland can have a disproportionately strong influence on aggregated indices. This is most noticeable where a small wetland completely disappears, giving an annual change ratio approaching zero. Therefore a filter was applied whereby any change greater than a halving or doubling in one year in a single wetland was taken out of the calculations. The filter removed some annual ratios from three Asian and one European marine/coastal wetland that more than halved each year and one Asian and one European marine/coastal wetland that more than doubled each year (see Table A2). All other annual area changes in the remaining wetlands fell within the range of halving to doubling.

The sub-indices and regional indices (I) were calculated relative to a standard baseline value equal to one in 1970:

$$\begin{aligned} I_y &= I_{y-1} \cdot \Delta_y \\ I_{1970} &= 1. \end{aligned} \tag{3}$$

We calculated 15 regional indices, comprising five marine/coastal indices, five inland indices, and five indices for all natural wetlands combined. Wetlands recorded as human-made in the database were excluded for the purposes of this analysis. As the database had time-series for only three natural wetland classes in the Neotropics (intertidal wetlands, permanent shallow marine waters and marshes on peat soils), robust indices for the region could not be calculated. Ninety-five percent confidence intervals for each index were calculated by a bootstrap method following Loh et al. (2005) using ten thousand bootstraps.

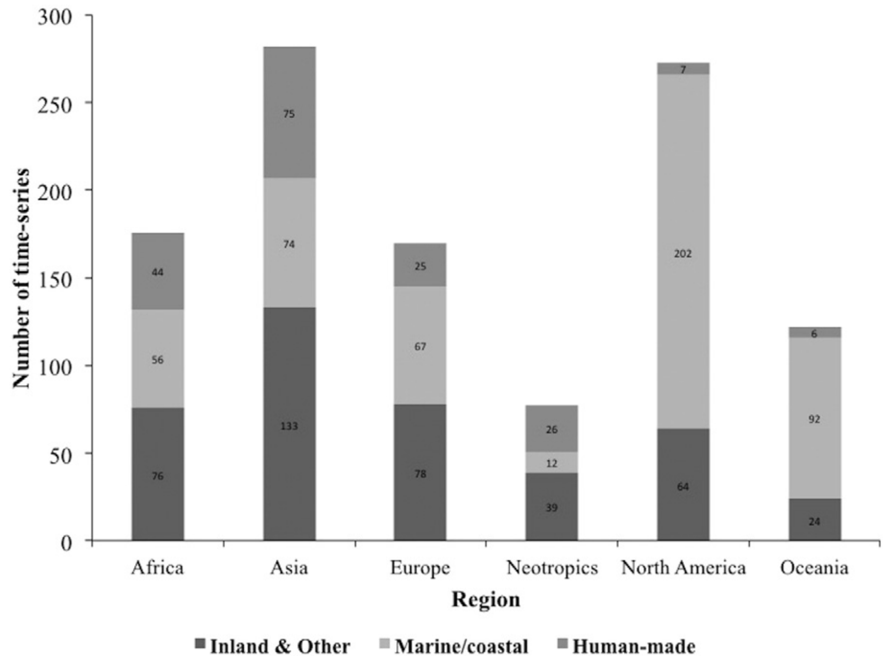

Fig. 1. Number of time-series records per region by wetland type.

2.6. Weighting and aggregation of the global indices

The calculation of the global WET indices departed from the method for calculating the global LPI in that unequal weightings were used to aggregate the regional indices. This was to compensate for the fact that the world's wetlands are unevenly distributed amongst Ramsar regions (Finlayson and Spiers, 1999). Weightings were derived from estimates of the total area of wetland in each region obtained from the Global Lakes and Wetlands Database (GLWD) (Lehner and Döll, 2004). The regional and type totals were recalculated from the GLWD using Ramsar regions and types (see Table 2).

The global marine/coastal, inland and natural WET indices were calculated by weighting the regional indices as follows:

$$I_{gy} = \prod_{a=1}^{n} I_{ay}^{\,w_a} \quad (4)$$

where I_{gy} is the global index in year y, I_a is the index for region a, n is the number of regions and w_a is the fractional weight of wetlands in region a ($\sum w = 1$) according to the GLWD.

The hierarchy of aggregations used to calculate the sub-regional, regional and global WET indices is shown in Fig. 2.

2.7. Investigation of size effects on rate of change in extent

If small wetlands tend to change at a faster annual rate on average than large wetlands, then an over-representation of small wetlands in the database would result in the index declining faster than the total wetland area, and vice versa, and result in an overestimate or an underestimate in the rate of wetland loss. To assess this possible bias, we first analysed whether there was an effect of initial wetland size on annual rate of change in wetland extent across the whole database using linear regression. Second, we investigated how closely the WET index's measure of average rate of loss in wetland extent approximates total wetland loss. As the WET database consisted of many non-overlapping time series that could not be summed, we tested the method on a dataset for the Mediterranean region (Mediterranean Wetlands Observatory, 2014). The dataset comprised wetland areas for 214 Mediterranean wetland sites from three points in time – 1975, 1990 and 2005 – derived from remotely sensed satellite data obtained from the European Space Agency and the Globwetland II project. We applied the WET method to calculate an index for the Mediterranean and compared the results to values of total wetland loss.

Table 2
Fractional weights of wetlands in the five regions used in the calculation of the global WET Index (area in thousand km^2 in parentheses).

	Natural		Inland		Marine/coastal	
Africa	0.16	(1542)	0.16	(1468)	0.13	(74)
Asia	0.21	(2101)	0.19	(1776)	0.59	(325)
Europe	0.19	(1920)	0.20	(1829)	0.16	(91)
North America	0.39	(3887)	0.42	(3878)	0.02	(9)
Oceania	0.04	(401)	0.04	(348)	0.10	(53)
Total (excluding Neotropics)	1.00	(9851)	1.00	(9299)	1.00	(552)

3. Results

3.1. Regional and global natural WET indices

The regional and global (minus the Neotropics) natural WET indices are shown in Fig. 3. The global natural WET index with regional GLWD-derived weighting declined by 31% between 1970 and 2008, with a 95% confidence interval (CI) of 28–33%. The ‘unweighted’ natural WET index, with equal weighting for each region, differed very little, also with a decline of 31% (CI 29-34%) over the same period.

When disaggregated, the regional natural WET indices show a decline in all five Ramsar regions assessed, with Europe showing the greatest decline (50%, CI 43–57%), and Oceania the least (17%, CI 10–23%) from 1970 to 2008. The marine/coastal WET index (weighted) declined by 38% (CI 35–42%) globally, faster than the inland WET index (weighted) which declined by 27% (CI 24–30%) over the 38-year period. Europe showed the most rapid declines of all regions in both its marine/coastal and inland indices. Percentage declines for all 15 regional and three global indices from 1970 to 2008 and the number of time series used in calculating each index are shown in Table 3.

Fig. 2. The aggregation of the individual wetland time-series to the Wetland Extent Trends (WET) index. Arrows denote taking the geometric mean of the level below, while the dotted arrows and bracket denote the application of weightings during the aggregation. Natural WET indices are an aggregation of inland and marine/coastal indices.

3.2. Investigation of wetland size effects

The global WET dataset exhibited a very small effect of wetland size on annual rate of change in area, with smaller wetlands experiencing faster rates of decline than larger ones (R-squared 0.0079, slope 0.0019, see Fig. A1 in the online Appendix). However the size distribution of wetlands in the database is close to normal. We acknowledge that beyond size alone, there could also be other factors not analysed that could contribute to the association of wetland size and rate of change in area.

The comparison using the Mediterranean dataset revealed that the trend calculated using the WET index method differed little from the absolute change in total area (Fig. 4). The Mediterranean WET index value in 2005 was 0.88 compared with a baseline of 1.0 in 1975, indicating a 12% decline (CI 8–16%) in wetland extent from 1975 to 2005. This decline compares with an actual decline in total area of approximately 10%, a value within the confidence limits for the Mediterranean WET index (Fig. 4).

4. Discussion

By modifying a method designed for species population trends, the Living Planet Index (LPI), we developed a global indicator of natural wetland extent trends based on the incomplete and uneven time-series data on wetland area available in the literature. An approach such as this is straightforward and relatively cost-effective to maintain and update. The results of the WET index indicate that, on average, natural wetlands declined by around 31% from 1970 to 2008, with marine/coastal wetlands declining more than inland wetlands.

There are inevitable caveats concerning our analysis. As with all meta-analyses, its quality depends on the completeness and accuracy of the case study descriptions upon which it is based. The data sources used a variety of methods to measure change in wetland area. Over half used Landsat images (e.g., Han et al., 2007; Li et al., 2013; Pattanaik and Prasad, 2011), including MultiSpectral Scanner (MSS), Thematic Mapper (TM) and Enhanced Thematic Mapper Plus (ETM+) satellite images, while other used aerial photographs (e.g., Evers et al., 1992; Jeanson et al., 2014; Tamisier and Grillas, 1994) and historical maps (e.g., Godet and Thomas, 2013; zu Ermgassen et al., 2012). Many sources used a mixture of methods in order to gather sufficient information to establish a trend and each method will have strengths and weaknesses. For example, while satellite imagery has advantages for assessing remote wetlands or those that would otherwise by prohibitively expensive to assess in the field, a major limitation is that infrequent imagery will fail to capture natural fluctuations in their area (Davidson and Finlayson, 2007). In order to maximise the volume of data used to calculate the WET index, no further judgement of quality was taken beyond filtering the data for extreme annual rates of change. There are also various possible sources of bias associated with the method.

A first potential bias relates to ecological and geographical representativeness. Data were taken from the literature according to availability, which is inevitably skewed towards better-studied wetland types and locations, as with the LPI (Loh et al., 2005). The wetland classes with less than 10 time-series globally were: for inland, Alpine tundra wetlands, Geothermal & subterranean and Mixed wetland type, and for marine/coastal, Mixed marine coastal wetland. Mixed wetlands are purposefully low in number as time-series were assigned a more specific class when possible. The wetland classes with the most time-series: for inland, Marshes on peat soil (208 time-series), and for marine/coastal, Permanent shallow marine waters (229 time-series) and Intertidal wetlands (214 time-series) (see Table A2). The over-representation of these wetland classes is due to the inclusion of existing data sets for these classes (FAO, 2007, 2014; Joosten, n.d.; Waycott et al., 2009). It is difficult to empirically assess the ecological representativeness due to the lack of a reference data set with similar wetland classes. As

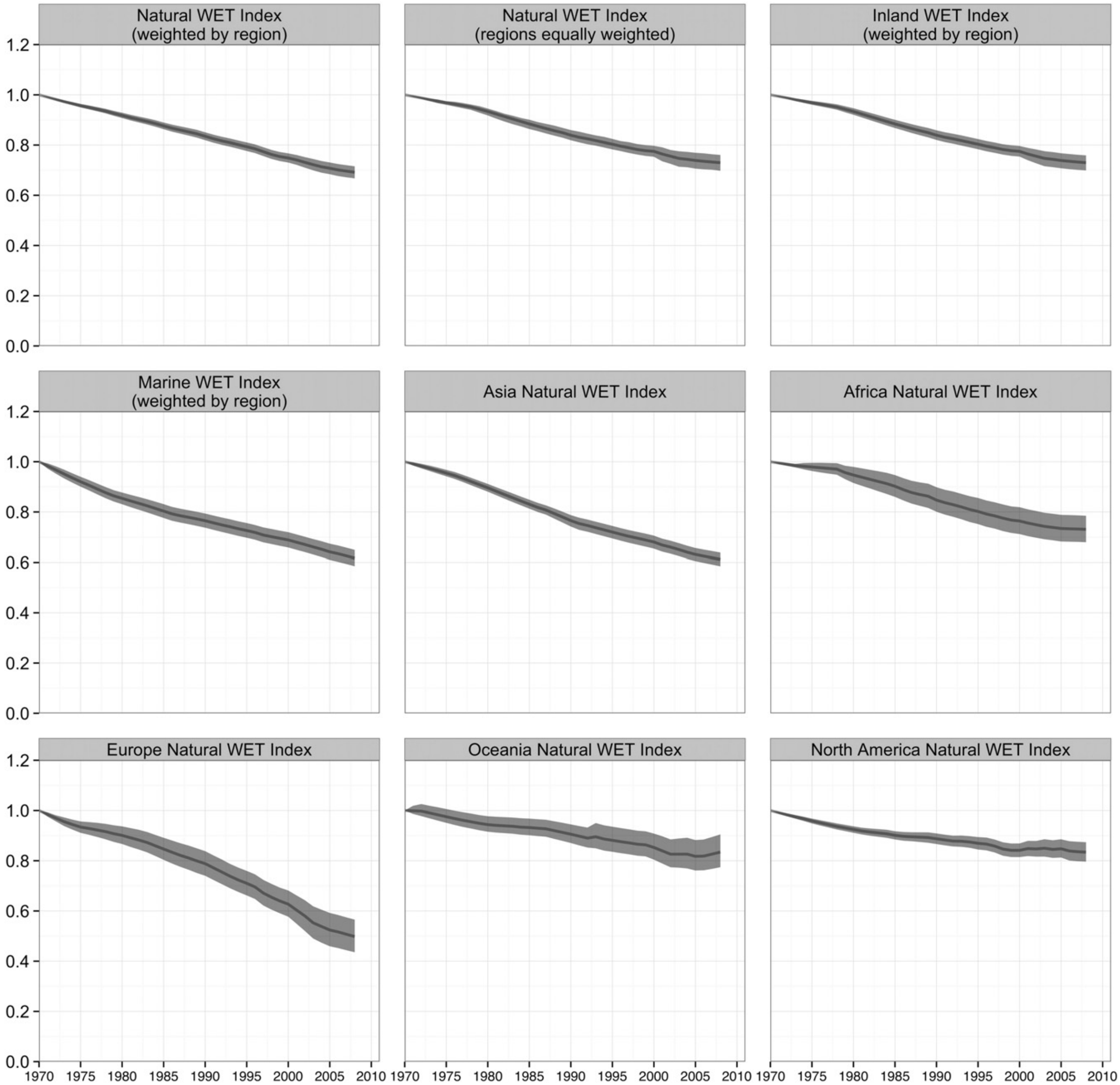

Fig. 3. Natural WET index weighted by region and with regions equally weighted, Inland and Marine WET indices and the regional natural WET indices (excluding the Neotropics) from 1970 to 2008 with 95% confidence intervals.

these data sets become available, the classes can be weighted accordingly. There are clearly regional data gaps. For example, it was not possible to calculate a regional index for the Neotropics, which has been recognised as being data poor for wetland extent data (Davidson, 2014; Finlayson and Spiers, 1999). As a result, the wetlands in the Neotropics are not currently represented in the global WET index. Efforts to fill such gaps can be made in the future by reviewing non-English scientific literature, administrative and NGO reports as well as collaborating with regional experts. Improvement to the WET index will rely on future expansion of the database to fill these gaps.

Table 3
Percentage declines from 1970 to 2008 in the WET indices (I), 95% lower and upper confidence limits (LCL and UCL) and number of time-series in each index (N).

	Natural				Inland				Marine/coastal			
	I	LCL	UCL	N	I	LCL	UCL	N	I	LCL	UCL	N
Africa	27%	32%	21%	132	31%	38%	23%	76	19%	26%	12%	56
Asia	39%	41%	36%	209	39%	43%	36%	135	41%	45%	37%	74
Europe	50%	57%	43%	145	51%	59%	43%	78	50%	59%	39%	67
North America	17%	20%	13%	266	4%	6%	1%	64	28%	34%	22%	202
Oceania	17%	23%	10%	116	15%	19%	11%	24	17%	29%	1%	92
Global (weighted, excluding Neotropics)	31%	33%	28%	868	27%	30%	24%	377	38%	42%	35%	491

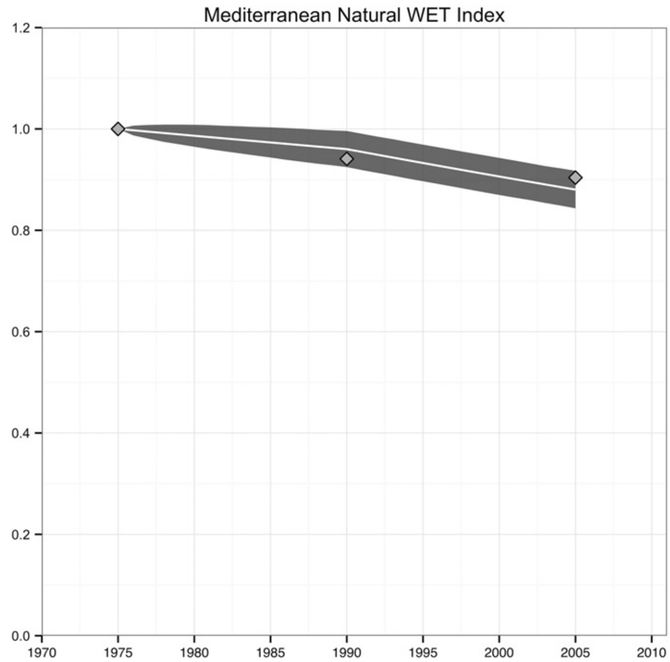

Fig. 4. Comparison between the overall trend calculated from the absolute change in total wetland area (diamonds) and the overall trend (with 95% confidence interval) calculated using the WET index method, for 214 wetland sites in the Mediterranean. Data provided by the Mediterranean Wetlands Observatory.

The WET index method attempts to address incompleteness and unevenness in the data by defining sub-region x wetland class matrices into which the data can be sorted, and then giving cells equal weight when constructing the regional indices regardless of the number of time-series in each cell. The WET index is therefore sensitive to where the boundaries are drawn between sub-regions and wetland classes. These boundaries are not as clear as the boundaries between species. This proof-of-concept WET index is intended to show how such a system of aggregating trends using such a matrix can work and make ecological sense. The greater the availability of time-series data, the smaller the sub-regions and more refined the wetland classes can be; the greater the number of data cells in the matrix, the more representative the resulting index. Given twice as much data, it would be possible to construct a more refined matrix and produce a more accurate and representative global WET index. Given complete global data coverage of all wetland types it would be possible to construct a matrix where the sub-regions are kilometre grid cells and the wetland classes cover every conceivable type, but then the necessity for an index which can use incomplete and uneven time-series data would disappear altogether, and a time-series of global wetlands could be constructed simply by adding up the total area in each year. Until such data are available, the WET index will be able to fill the gap in our knowledge as far as the available data allow.

To take some account of the inter-regional distribution of wetlands, the global WET index was aggregated from regional indices weighted using the wetland distribution statistics in the GLWD (Lehner and Döll, 2004). The GLWD, as other global wetland area assessments such as the 'Global review of wetland resources and priorities for wetland inventory' (Finlayson and Spiers, 1999), is widely recognised as underestimating some wetland types and regional wetland areas, especially for Africa, the Neotropics and Oceania (MA, 2005b). Although this source is not representative of a single point in time nor is it ground validated, the GLWD is well known and widely used. The effect of using regional weighting in the global WET index was not very different from giving equal weight to each region (see Fig. 3). According to the GLWD, Africa, Asia and Europe have comparable areas of wetland. The regions which differ markedly are North America, which has roughly twice the wetland area of Africa, Asia or Europe, and Oceania which has approximately a tenth that of North America. Using weighting therefore has the effect of making the global index closer to the North American trend and less like the Oceania trend. However, the regional WET indices for North America and Oceania are similar, hence the effect of weighting was small.

For future iterations of the WET index these weightings can be improved as better wetland distribution maps become available. Efforts to improve land cover mapping including wetlands from earth observation data are underway including the GlobWetland initiative of the European Space Agency (Jones et al., 2009) and various applications of Landsat imagery (e.g., Cao et al., 2014; Jun et al., 2014).

Another type of bias would occur if published studies were conducted on wetlands known to be particularly threatened or undergoing dramatic change in area. This type of selection bias would lead to a possible exaggeration of rates of wetland decline and might be a particular risk if the database largely comprised studies of individual wetlands. However, the database includes studies conducted at a range of scales from individual wetland sites to studies of multiple sites across a region. Larger scale studies often used remotely sensed data that is less likely to be selective and therefore further reduces the risk of bias. Future expansion of the database from a wider range of sources (e.g., national reports) may also help to address this source of bias.

Although selection bias cannot be entirely ruled out, the global result of approximately 1% annual average decline in natural wetland is within the range of results (0.34–2.1%) from a selection of large-scale remote sensing studies of wetland or surface water extent over the same period or a subset thereof (Gong et al., 2010; Mediterranean Wetlands Observatory, 2014; Niu et al., 2012; Prigent et al., 2012). Those with lower rates of loss include artificial wetlands that are increasing in extent — the results for natural wetlands alone would be higher and therefore closer to the WET result. This suggests that the WET index is not being heavily influenced by selection bias. It may in fact be partially offset by the interpolation method used in the creation of time series. The method assumes a constant rate of change in area over time to interpolate missing annual extent values. However, the rate of loss of many wetland classes is likely to have accelerated in the 20th century (Davidson, 2014). We included time-series with historical baselines prior to 1900 and only one or two recent data points. Because the constant annual rate of change was applied over a century or more, these time-series have a flattening effect in the recent period covered by the WET index (1970–2008). An accelerating annual rate of decline in the interpolations might have been more realistic, but in the absence of intermediate data points we assumed a constant rate.

A further source of bias might arise if rate of change in wetland extent is influenced by wetland size and the size distribution in the database is skewed. Our analysis indicates that wetland size has little effect on rate of change in area, and the trends in average change and total change in Mediterranean wetlands are very similar. This is important because a possible misinterpretation of the indicator could occur if the WET index's measure of the average rate of loss differs significantly from the absolute total area lost. By exploring this relationship with a dataset for the Mediterranean region for which absolute area change was known, we have illustrated that the WET index is robust to such misinterpretation.

Our findings lead us to believe that some of the principal potential biases that might affect the WET index have been minimised, and can be reduced further primarily by the inclusion of additional data.

We believe that the WET index is policy-relevant. Like the LPI it is intuitively simple, cost effective and can be updated regularly as well as disaggregated ecologically and geographically. It is directly relevant to two of the Aichi Biodiversity Targets adopted by Parties to the CBD in 2010. In particular it provides a baseline rate of wetland loss for Aichi

Target 5: to halve the rate of natural habitat loss by 2020. With the inclusion of more recent data as they emerge, the WET index would be able to determine whether the rate of wetland loss is declining compared with the current pre-2010 baseline. It is also relevant to the Priority Areas of Focus in Ramsar's 2016–2024 Strategic Plan (Ramsar Convention, 2015), stated as "Preventing, stopping and reversing the loss and degradation of wetlands", which speaks to the original 1971 desire of the Contracting Parties to "stem the progressive encroachment on and loss of wetlands now and in the future" (Ramsar Convention, 1971). While in the current analysis the focus is on natural wetlands, the Ramsar Convention also includes human-made wetlands, indices for which could be created in a future analysis. At the regional scale, assessment of wetland areas and ecological status are done in a few well-monitored regions (see for instance the Great Lakes Network Monitoring & Inventory Program, https://greatlakesmonitoring.org, or the European assessment EEA, 2015), but there is a lack of wetland inventories and assessment of their change in area in many regions. This was highlighted in the last Conference of Parties of the Ramsar Convention, where it urged for the development of regional initiatives and monitoring structures (Ramsar Convention Secretariat, 2015a). The regional WET indices could help fill this knowledge gap. Both the CBD and Ramsar Secretariats have used preliminary unpublished results from the WET index in recent promotional material and assessment products and both are interested in its robustness as a potential global indicator (Gardner et al., 2015; Leadley et al., 2014; Ramsar Convention Secretariat, 2015b).

More broadly, the WET index is relevant to at least three of the targets under the Sustainable Development Goals agreed by the UN General Assembly in September 2015, for which indicators will be agreed in early 2016. The UN Statistical Division has compiled a proposed list of indicators for which candidate metrics are being sought and evaluated, including an indicator of wetland change (indicator 6.6.1: percentage change in wetland extent over time).

As a measure of change in area, the WET index does not reflect change in ecosystem condition, which is an equally important characteristic of wetlands and other ecosystems (Ramsar Convention, 2005; Pereira et al., 2013). A separate indicator would be required to monitor wetland condition. Nevertheless the results of the WET index imply that, globally, the objective of the Ramsar Convention to ensure conservation and wise use of wetlands is not being fully achieved, whilst achievement of Aichi Target 5 would require a reduction in the global annual rate of loss to less than 0.5%.

Finally, if the WET index method works successfully for measuring wetland extent trends then we believe the same method could be used to create other ecosystem extent indicators, thereby filling this recognised indicator gap.

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.biocon.2015.10.023.

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