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Long-term trends in the abundance of Mediterranean wetland vertebrates: From global recovery to localized declines

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ABSTRACT

Biodiversity loss is unevenly distributed in space and time. Species have reached critically low population sizes in some areas, and remain abundant in others. Similarly, some species may benefit from successful conservation plans, while others still experience severe population depletions driven by negative impacts of human activities. Although several indicators have been proposed to measure the fate of biodiversity, they are generally only implemented globally so their relevance for regional assessment is still unclear. Here, we calculated the first regional trend in the Living Planet Index for the Mediterranean wetlands (Med LPI), an indicator that summarizes the fate of global biological diversity based on the temporal trends in abundance of vertebrate populations. The Med LPI was based on 1641 vertebrate populations of 311 species recorded in Mediterranean wetlands from 1970–2008, in 27 different countries. We investigated whether trends in the Med LPI differed between eastern and western Mediterranean countries, which have different socio-economic contexts. Finally, we assessed whether and how the trend in the Med LPI was robust to changes in the number and identity of species considered. We found that, at the Mediterranean scale, the Med LPI increased steeply, which could be taken at first sight, as a general recovery of wetland biodiversity in this biogeographical region. However, we found highly contrasting spatial trends within the Mediterranean region: the average trend was positive for western and negative for eastern countries. Moreover, we showed that depending on the method used to estimate the trend in Med LPI, it can be sensitive to the number and identity of the species considered. We suggest that understanding the regional discrepancies of the trend in biodiversity indicators as well as their robustness to the species represented in the index will enhance progress assessment towards global and regional conservation strategies.

1. Introduction

Most governments have identified reducing the rate of biodiversity loss as one of their global priorities (the CBD 2010 target, Balmford et al., 2005a). Large-scale conservation assessments are thus necessary to provide a general perspective on the current and projected status of biodiversity (Whittaker et al., 2005; Brooks et al., 2006). Hopefully, global conservation strategies have recently benefited from the increasing number of studies investigating spatial patterns in biodiversity, helping the evaluation of global conservation priorities (Ferrier et al., 2004).

However, measuring large-scale and long-term trends in biodiversity remains a challenging issue for two main reasons. First, the range of taxa and biomes covered by standardized monitoring schemes are incomplete and biased (Balmford et al., 2005b). In particular, trends in biodiversity are mainly based on records of temperate rather than tropical areas (Collen et al., 2008), on the fate of the terrestrial megafauna and megaflora, and on few taxonomic groups. Second, biodiversity is multifaceted and there is a common agreement that no single biodiversity indicator will ever summarize the fate of biodiversity and its relationship with human pressures (Purvis and Hector, 2000). Although the relevance of any biodiversity indicators obviously depends on the question being addressed and on the data available (Baillie et al., 2008), robust and consensual measures of progress towards the 2010 target are still missing (Walpole et al., 2009).

Among available biodiversity indicators, aggregated indices of multi-sites and multi-species trends have become one of the best available proxies for measuring trends in biodiversity (Balmford

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et al., 2003). In this respect, the Living Planet Index (LPI) has been developed to provide policy-makers, scientists, and the general public with information on trends in the abundance of vertebrate populations from around the world (Loh et al., 2005). The LPI was shown to be a heuristic instrument reflecting the fate of global biodiversity, but also in which ecosystems species are declining most rapidly (Collen et al., 2008).

The strength of such composite indices is that they are simple to communicate while providing sensitive measures of biodiversity change, and flexible enough to handle data from different sources, collected with a variety of methods, and at several scales (de Heer et al., 2005). The LPI was thus adopted as a key indicator of the state of global biological diversity at the international level, and is one of the indicators selected by the Convention on Biological Diversity as a tool for measuring progress towards the 2010 target (Balmford et al., 2005a). However, research has principally focused on species declines and/or extinctions either at site level, local or very global scales, while the comparisons of biodiversity's fate across different biogeographical contexts of land-use, climatic, and conservation history are far less common.

In this respect, assessing the long-term trend in abundance of vertebrate populations in Mediterranean wetlands is particularly relevant for several reasons. First, the Mediterranean region represents major biogeographic cross-roads in which the paleogeography and historical land-use have created a complex mosaic of habitats with high species richness, high spatial turnover in biodiversity and high rate of endemism (Médail and Quézel, 1999). Second, the Mediterranean and Black Sea wetlands are experiencing increasing pressure from human activities such as urbanization, tourism, changes in natural flood regime, drainage, pollution, and agricultural intensification (Underwood et al., 2009; Eglington et al., 2008). This has led the Mediterranean basin to be recognized as one of the first 25 Global Biodiversity Hotspots (Myers et al., 2000). Third, as Mediterranean wetlands are physically and socially connected, ecological processes occur over a much wider territory than a particular site or country (Amezaga et al., 2002).

Assessing the general trend in biodiversity in the Mediterranean wetlands should help to justify and stimulate better local conservation targets and improve short and long-term global land-use planning (Spector, 2002). The method used to create the global LPI can also be applied at sub-global scales to measure trends in the status of biodiversity for any given set or group of species and/or any geographic scale, provided that sufficient data are available. Aggregated trend indices have thus been generated for biogeographic realms, biomes, habitats, for particular taxonomic groups, and for certain countries (Collen et al., 2009; McRae et al., 2008). Yet, an overall biodiversity assessment is missing for the Mediterranean wetlands for which only the fates of particular groups and/or a particular country or site have been investigated (e.g. Moroccan waterbirds, Green et al., 2002). Therefore, comparing trends in biodiversity among different biogeographic regions within the Mediterranean basin may be useful to discern between different drivers of biodiversity change. Moreover, whether the general trend in such a biodiversity indicator is sensitive to the particular number and identity of species considered has not been explicitly tested.

Here, we address four main objectives: (1) investigating the long-term trend in the LPI for Mediterranean wetlands considered as a whole, (2) assessing whether the general trend is similar at sub-regional levels among eastern and western Mediterranean countries. (3) Regardless of any trends, some regions can be of critical conservation importance in being the stronghold for many species. We thus also compared the eastern versus western population sizes of species present in both regions. (4) Finally, we tested the robustness of the trend in the Med LPI to changes in the species considered.

2. Methods

2.1. Population data

We collected time series of total population numbers or proxies for estimate of population sizes (e.g., counts, density measurements, biomass, and number of nests) and taken from different sources. To be included in LPI calculation, data had to fit a number of criteria, following Collen et al. (2009). In brief, population size must be monitored for at least 2 points in time, the geographic location of the population as well as the data source must be available, and data must have been collected using the same method on the same population throughout the time series. We collated vertebrate population time-series published in scientific journals, or in scientific reports, books and websites of non-governmental organizations and focused on data for which a special standardization effort had been made explicit (e.g. population surveys, restoration programs, protected area assessment). We stopped collecting data when the inclusion of new time series was negligible compared to those already collected.

We collected time series for 1641 georeferenced vertebrate populations with at least two data points between 1970 and 2008. These populations covered 311 vertebrate species (among which 247 bird species, 36 freshwater fishes, 12 amphibians, 8 mammals, and 8 reptiles). These species were monitored in 27 countries distributed across the Mediterranean basin including Black sea (Fig. 1). Although still incomplete (overall, 1245 vertebrate species are known to occur in the Mediterranean basin), our dataset include the more well-monitored species of different taxonomic groups which have a range of ecology and sensitivity to human impact. Therefore, we believe that this dataset provides a good synthesis of the trends in major vertebrate populations currently monitored in the Mediterranean wetlands.

2.2. Calculating aggregated trends in abundance

Two alternative methods were used to calculate a trend in the Med LPI: (i) the chain method (Loh et al., 2005), with generalized additive modeling techniques (GAM) (Collen et al., 2009) and (ii) linear mixed modeling. In the first approach, the trend in the index is estimated following three steps (see Collen et al., 2009 for more details). First, for each species, the average change in population from each year to the next was calculated. In case of incomplete time-series, zero values were first replaced by one percent of the mean population measure value for the whole time series. Then, the population sizes for these years were derived from interpolation of the preceding and the subsequent years with measured values (see Collen et al., 2009; Craigie et al., 2010) as follows. For time series with six or more population measures we implemented Generalized Additive Models (GAMs), following Collen et al. (2009). GAMs are extension of classical linear models in which the predictor is not restricted to be linear but is the sum of smoothing functions. For long time-series, this smoothing approach better describes non-linear temporal variations in populations' sizes. Those time series with five or fewer population measures were estimated through log-linear models.

Second, the trends of all populations of the same species were averaged to produce species-specific trends. Third, the average rate of change in each year across all species was calculated. Finally, the average annual species change in each year was chained to the previous one to make a continuous index, starting with an initial value of 1 in 1970. Note that in these approaches, equal weighting are given to each species within the index. Precision of the estimates was accounted by means of 95% confidence intervals generated using a bootstrap technique in which 1000 index values are

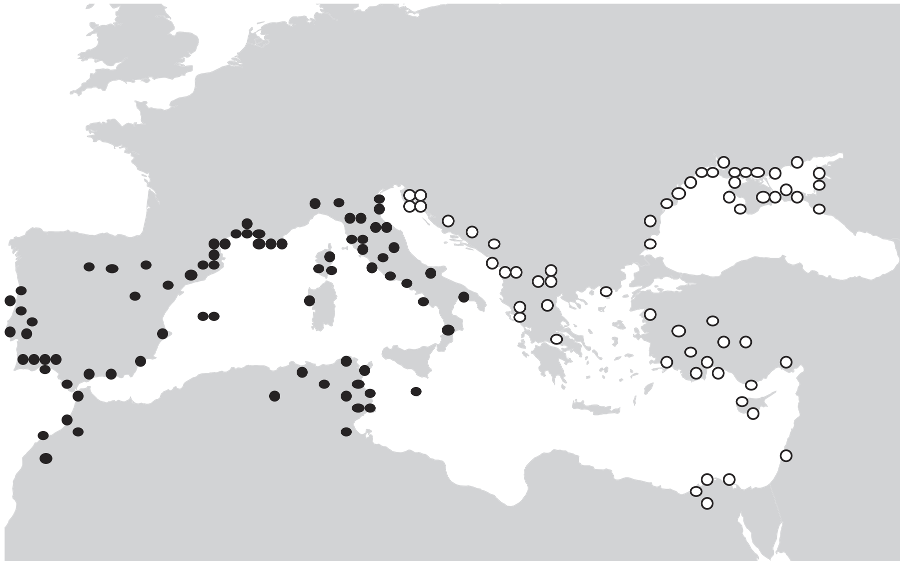

Fig. 1. Spatial distribution of major monitored sites (black and white plots are monitored in western and eastern Mediterranean countries respectively).

calculated each year from species-specific population changes randomly sampled (Loh et al., 2005).

We also calculated Med LPI using a classical regression model. Using regression is a more conservative statistical approach as all the data are considered in one general model and that no interpolation is required for discontinuous time series. All the inter-and intra-species variability is thus considered to estimate annual changes in LPI. In this model, log-transformed population size (or estimate) was considered as a dependent variable, species and year as main independent factors, and site as a random effect. This analysis provided estimated coefficients β_t of the index for each year (and associated standard error, $\beta_t$se) representing differences in overall abundance values between each year and the reference year (1970), accounting for differences among species. In this analysis, the relative contribution of a given species' trend to the index is dependent on the number of populations monitored for that species. The analysis was therefore weighted by the number of populations n_i monitored for each species (using $1/n_i$, as a weight) (Loh et al., 2005).

To estimate an overall linear trend in the aggregated index, we performed a regression model using estimated coefficients β_t as the dependent variable (or each year-to-year rate of change provided by the chain method) and year as a continuous variable. This latter model was weighted by each coefficient standard error to account for the difference in confidence of each estimate (using $1/(\beta_t\text{se})^2$ as a weight).

Note that in the LPI calculation, a population can be monitored at a specific site (e.g. pond, river or Natural Park) or across a large geographical area (administrative region, country). In other words, no minimum population size was required for selecting time series so that small and large populations had the same weight in the index calculation. However, raw abundance data were treated as the log differences in population sizes. In doing so, all trends obtained with the chain method reflected changes in relative (not actual) abundance either within species (when lambda values are averaged across the populations of a given species) or between species (when species-specific lambda values are combined together). Using regression, the population was treated as a fixed factor, which accounts for the difference between populations' sizes. To check for possible influence of highly stochastic dynamics of very small populations, we also calculated the index using only populations that represent more than 1% of the Western Palearctic population of the species concerned and found similar results (not reported).

We further tested whether the general trend in the index could mask regional heterogeneity. In particular, we expected the fate of biodiversity of eastern and western European countries to be influenced by different biogeographical and socio-economic history and also by more recent differences in conflicts between the conservation of biodiversity and other human activities (Young et al., 2007). Therefore, we assessed the trend in the index for the eastern countries only (Fig. 1), to shed light on possible uneven spatial distribution of trends in LPIs of Mediterranean wetlands (892 populations were monitored in the west and 749 in the east).

Finally, we also tested (using a paired *t*-test) whether population sizes differed between eastern and western Mediterranean wetlands. We selected species for which recent population size estimates were available for both regions ($n = 72$). For these species, population sizes were taken from similar data sources and estimated with similar methods (Birdlife international, 2004).

2.3. Testing index robustness

We assessed whether the trend in the Med LPI was dependent on the number and identity of the species considered using a systematic approach in which the index was calculated after the removal of species randomly selected from the initial species pool. We varied the number of species randomly removed from 1 species to 80% of the initial total of 311 species (i.e. for a removal of 90%, the index was calculated on 31 remaining species) and in each case, the index was calculated for 10,000 different sets of those species randomly selected. From this analysis, we assessed the robustness of the index to change in the species considered in drawing (i) the trends obtained when 30% of the species were removed from the initial pool and (ii) the continuous relationship between the average and the minimum of the 10,000 trends obtained from the simulation, and increasing percentage of species removal. We performed this robustness test for the chain method and the regression model.

3. Results

The dataset contained a high proportion of bird species (79% of the species considered) and data were not equally distributed among Mediterranean countries, reflecting a great heterogeneity of data collection and/or publication among taxonomic groups and geographic areas. Using the regression method, we found a steep increase in abundance for the overall index of Mediterranean wetland populations, (slope: 0.04, $F_{1,38} = 299.99$; $P < 0.001$). The overall Med LPI has almost tripled in four decades (Fig. 2a). In contrast, we found that the Med LPI for the eastern Mediterranean region decreased over this time period (slope: −0.008, $F_{1,31} = 5.27$, $P = 0.02$, note that for eastern countries data were available for the period 1970–2002 only) (Fig. 2b).

Finally, among species found in both eastern and western Mediterranean countries, population sizes were much higher in the east (paired *t*-test performed on population sizes for the 72 species present both in eastern and western countries and taken from similar data sources: $t = -2.80$; $P = 0.006$).

We then calculated temporal trends in abundance for different subsets of species randomly selected from the initial pool. For instance, when only 30% of the species were removed from the initial species pool, we found that using the chain method, the resulting general trends in the Med LPI were highly dependent on the identity of the selected species, resulting in relatively wide confidence limits (upper and lower slope in LPI: 0.04 and −0.006 respectively; Fig. 3). We repeated this analysis with varying proportions of species removed from the initial pool and found that, on average, the trends were not highly affected by the removal of few species (Fig. 4a). Indeed, the average trend in abundance resulting from the removal of less than ~20% of species from the initial pool did not differ from the original trend. However, following a greater proportion of species removal, the average trend calculated with the chain method increased exponentially (Fig. 4a). Moreover, focusing on the minimum of the temporal trend in the index (rather than on the average), this analysis reveals that after the removal of few species only (~10%), negative trends in the index could result from this species removal (Fig. 4b).

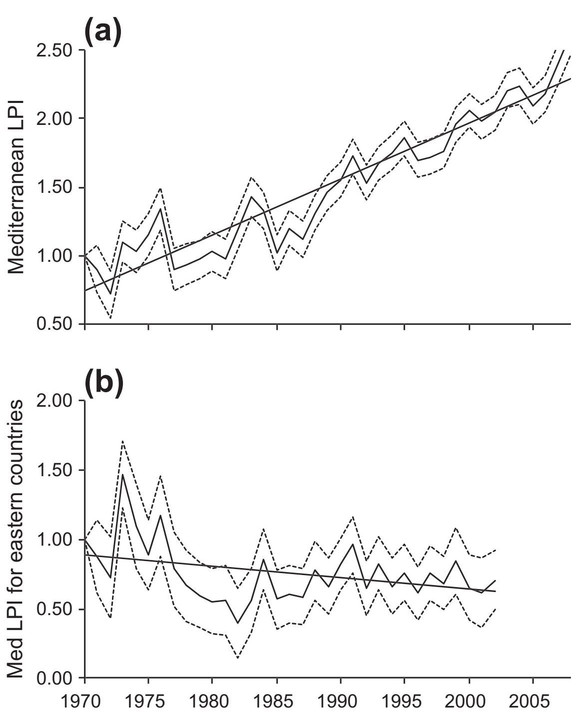

Fig. 2. Spatial trend of the Living Planet Index obtained from the regression technique for (a) Mediterranean wetlands (1970–2008) (dashed lines represent ± SE around the trend) and (b) for data collected in eastern countries only. Note that not enough data were available to calculate the trend in LPI beyond 2002 for eastern countries.

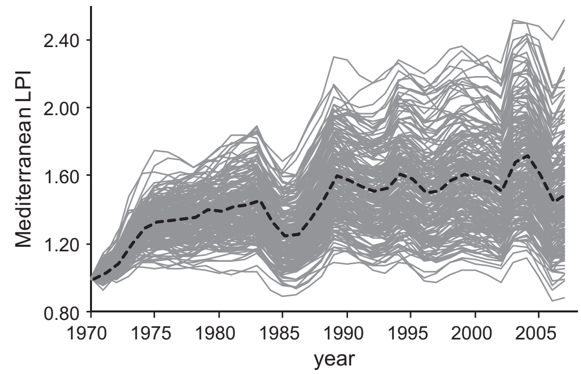

Fig. 3. Living Planet Index trajectories obtained after the removal of 30% of the species from the initial pool. The dashed line represents the observed trend calculated with all species.

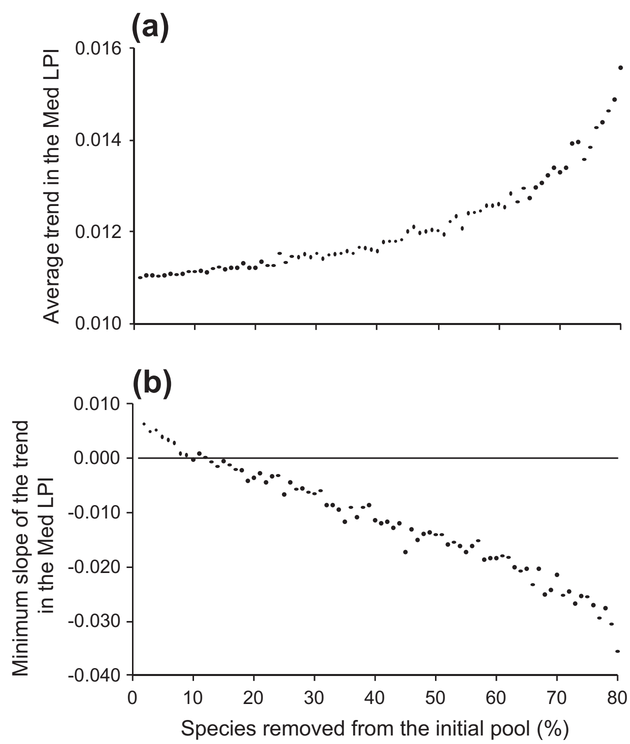

Fig. 4. Relationship between (a) the average and (b) the minimum slope of the temporal trend in LPI (*Y* axis) calculated on 10,000 trajectories and the number of species removed from the initial pool (*X* axis, expressed as a percentage). The horizontal line delineates positive and negative minimum values.

A similar robustness test conducted on the regression model suggests that the estimated trend in LPI is even more robust to changes in the identity of the species considered using this approach. Indeed, regardless of the number of species considered, the average trends were always positive and very close to the one obtained with the complete pool (Supplementary Fig. 1).

4. Discussion

4.1. Contrasting trends in LPI for Mediterranean wetlands

Overall, we found a strong increase in the Med LPI between 1970 and 2008 for Mediterranean wetland species. This positive trend may be surprising given the negative fate generally reported for biodiversity, and more specifically, given the acknowledged negative impacts of global changes on freshwater biodiversity (Dudgeon et al., 2006). The encouraging signal found in this study is most probably partly driven by the progressive recovery of particular species positively affected by more sympathetic landscape planning in the region. In particular, the European Birds Directive (1979) may have had positive effects on species dynamics of conservation concern. Supranational conservation policy was indeed recently shown to bring measurable conservation benefits on bird populations (Donald et al., 2007).

Moreover, a close inspection of the data revealed a recent re-colonization of species in the west, previously limited to the eastern part of the Mediterranean Basin. These local increases can result from the colonization of individuals from populations located in the eastern and southern part of the Mediterranean basin and/or from the local change in environmental conditions.

For instance, negative impacts of land-use changes in the east have been cited as driving forces of the transfer of individuals for the Mediterranean Gull *Larus melanocephalus* and Slender-billed Gull *Larus genei* that settled in the North-western Mediterranean in the mid-1980s (Sadoul, 1997). Glossy Ibis *Plegadis falcinellus*, European Spoonbill *Platalea leucorodia*, and Squacco Heron *Ardeola ralloides* are other waterbird species that have shown a dramatic and recent increase of their breeding numbers in the Western Mediterranean whereas severe declines were recorded in their former Eastern strongholds like Romania and Turkey (Birdlife International, 2004). The expansion to the west of these species may be the consequence of conservation problems met in their traditional breeding grounds in the east.

However, there are evidences that some waterbird species really improved their conservation status in the Mediterranean independently of any transfer of populations. The Great crested Grebe *Podiceps cristatus*, the Common Shelduck (Tadorna tadorna) or the Cattle Egret *Bubulcus ibis* are just few examples of species that increased in the west and remained stable in the east (Birdlife International, 2004). Among other causes, stricter hunting control (no hunting during the pre-breeding migration and nesting periods) together with the adoption of specific management measures may have enabled water bird populations to recover (Donald et al., 2007). The banning of some particularly dangerous pesticides (e.g. DDT) in the 1970s in many countries also positively influenced the species situated at the top of the food chain (Sakellarides et al., 2006). These factors probably contributed to the highly positive trends of species previously persecuted by hunters and fishermen and recently protected (e.g. Grey heron, *Ardea cinerea*, +230% in 34 years) and of the rapid increase of formerly uncommon species (e.g. Yellow-Legged Gull *Larus michahellis*, +256% in 36 years, Little Egret *Egretta garzetta* +202% in 39 years). Further changes in waterbird communities can be still expected in the west as suggested by recent increase in the settlement of species that were previously restricted to the eastern Mediterranean (e.g. the Pygmy Cormorant *Phalacrocorax pygmeus* which is currently spreading to Italy from the Balkans).

However, beyond this general increase, we found that the positive trend in Mediterranean vertebrate species abundance was not equally shared by western and eastern countries, with a negative trend in the eastern region. The difference between western and eastern regional indices likely reflects the recent economic development of the northeastern region combined with strong human demographic growth and increasing pressure on water resources mostly in the southeastern region. The production and use of energy in east-central Europe associated with the recent development of eastern countries has had detrimental effects (either directly or indirectly) on upon surface water quality. While significant amount of eastern European wetlands has been converted to agricultural use in the past, remaining wetlands are still subject to rapid and recent agricultural drainage (Hartig et al. 1997). Moreover, freshwater consumption in most countries of the south-eastern Mediterranean region now exceeds renewable resources (Wackernagel et al., 2006), and the rapid development of agriculture, manufacturing industry and tourism was recently followed by the disappearance and deterioration of many wetlands. More localized wetland transformations have also affected eastern wetlands recently. For instance, a site of major importance for 6 species of threatened colonial waterbirds (Drana lagoon) was destroyed in 1987 in Greece for the development of farming (Goutner, 1997). Similarly a pronounced increase in Turkish croplands took place between 1972 and 1987 at the expense of the irreversible losses of Lake Amik and its related wetlands of over 53 km^2 (Kilic et al., 2006). These rapid land-use changes have probably contributed the negative trend observed in the eastern Mediterranean LPI.

As with all biodiversity indicators, the baseline year and start of the index is critical. For example, the partial recovery of the populations observed in western countries is likely to have started from depleted levels in 1970 as most countries of the western Mediterranean basin underwent early development during the 19th century, and that the majority of wetland drainage impacted western wetland biodiversity before the 1980s (Smart et al., 2006). In contrast, the negative trend in the eastern Mediterranean may represent important population depletions, as this region is a key area hosting most of the breeding populations of many species of water birds (Brinson and Malvárez 2002).

This result suggests that any indices based on variations of relative population sizes should be coupled with analysis of real abundances. Here, while the very strong positive trend in the western Med LPI results from the increase of relatively small number of individuals; the smaller decrease in the east corresponds to the loss of large number of individuals.

4.2. Limitations of the LPI approach to assess the fate of biodiversity in Mediterranean wetlands

The LPI is a measure of trends in global vertebrate abundance. The extent to which this is representative of wider trends in all species, and ecosystems remain unknown (Loh et al., 2005). For instance, the data on which the index is based are combined from studies undertaken for diverse motivations and it is likely that species known to be threatened are generally better monitored than other species. Further investigation is required to devise to what extent the trends indicated by time-series derived from data available for some species are representative of the fate of all vertebrate species, and more generally of the trend in biodiversity in the biogeographical realm they occupy (Lughadha et al., 2005).

In this study, the general trend for the Med LPI is subjected to bias in the spatial distribution of the data used. Almost 85% of the data used came from the northern part of the Mediterranean Basin. Unfortunately, this bias reflects data availability, due to the difficulty of accessing data collected in southern countries, and/or the low number of monitoring programs in some areas. For example, there is no regular monitoring of wintering avifauna in the most important Mediterranean wetland, the Nile Delta (Meininger and Atta, 1994). Yet, it is precisely in these regions that wetlands appear to be the most threatened, as those countries have experienced recent economic growth and conservation targets are not perceived as a priority yet (Young et al., 2007). Although other spatial grouping of vertebrate time series (e.g. northern versus southern countries) could be very interesting to compare the resulting trend in LPI, they will be blurred by uneven sampling effort among groups. Overall, we suggest that the Med LPI does not completely mirror what is happening in reality in this region and that increased monitoring efforts are required in the southern section of the Mediterranean basin.

Similarly, the Mediterranean LPI was biased towards few taxonomic groups and the positive trend in the west to some extent is a reflection of the trends of gregarious and charismatic water birds, which are easier to count, or for which specific surveys exist. Other classes of vertebrates (amphibians, reptiles, fish and mammals) are therefore under-represented (only 20% of the data considered). This bias also confirms that Mediterranean freshwater and marine biodiversity has received only a fraction of the attention accorded to its terrestrial counterpart (Bianchi and Morri, 2000). To account for this bias, one could use *a priori* weighting (i.e., in calculating LPI within different strata of unequal sizes, see Loh et al., 2005). However, while it is reasonable to weight to account for different sampling efforts (e.g. between regions of different size), having different weights for different groups can be hazardous. Indeed, there is no particular reason to remove the differences in species numbers within different groups when this difference is real. What is more important however is to ensure that LPI is based on representative datasets for each group. Therefore, better knowledge of the trends of the other taxonomic groups is essential to refine our findings. The global trend of fish, reptiles, amphibians and mammals suggests that these taxonomic groups are facing negative impacts of global changes (Stuart et al., 2004; Dudgeon et al., 2006), which is confirmed by recent assessment of conservation status. Initiating standardized monitoring programs focused on particular sites can be time-consuming and costly. So-called citizen-science programmes can be a good alternative (Devictor et al., 2010). Such programmes (e.g., Midwinter Waterfowl Census) are particularly successful in some wetlands all around the world but are still missing in other countries albeit important for wetland conservation (e.g. Egypt, Nile delta).

Finally, we showed that removing species from the initial pool can alter the general conclusions regarding the fate of biodiversity for the Mediterranean wetlands. For instance, removing 30% of species from the initial pool (i.e., calculating the index on 218 species instead of 311) can lead to very different trajectories (including negative rather than positive trends, Fig. 3). We also found that as the number of species removed from the initial pool increased, the average trend in LPI calculated on the remaining species increased. This pattern results from two different sources of bias. First, the variance of the average year-to-year change in population size increases as sample size decreases (a consequence of the central limit theorem, Rice, 1995). In other words, the probability to find combinations with extreme dynamics (with highly positive or negative trends) increases as fewer species are considered. This bias is illustrated by the decrease in the minimum of the trends of the index obtained following increasing species removal (Fig. 4b). Second, to calculate the Med LPI, the population change in each year is chained to the previous one using $I_t = I_{t-1}\ 10^{\bar{a}t}$, where $\bar{a}t$ is the mean of each species-specific year-to-year change in population size (Loh et al., 2005) (for a given population of a given species, $at = \log_{10}(N_t/N_{t-1})$). This calculation induces a non-linear relationship between the index and the variance of the species-specific arithmetic mean ($\bar{a}t$). As the variance of this mean increases with species removal, the number of combination including species with positive dynamics is higher which also positively inflates the overall Mediterranean LPI (Fig. 4a). This bias is therefore removed using a regression model instead of the chain method (Supplementary Fig. 1).

Substantial change in the average value of the index is only obtained above the removal of 30% species from the initial pool. However, even after the removal of a relatively small proportion of species (~10%), a negative index trend could be obtained. These results show that although the majority of species increased during the period considered, the inclusion or removal of few species can greatly influence the overall index. This test illustrates the importance of using many local population trends when generating regional and global indices of biodiversity.

We suggest that analysing this bias using such a randomization approach provides an interesting tool to diagnose robustness. Indeed, the shape and celerity of the trend in aggregated indices (such as LPI) with increasing species removal is helpful to assess whether and how the results are dependent on the species considered. The strength of this bias is different for each dataset and will depend on the quality and quantity of the data available. The degree to which this bias matters will then depend on (a) what statistical analysis is used, (b) what and how many species is absent from the dataset, and (c) how different their trends are to the species considered. Depending on the robustness assessment, one can decide to wait until more data are collated before producing a trend for a given region.

Note also that while the chain method and the regression model provided similar qualitative results, the quantitative estimates were different (the slope in global Med LPI was steeper when estimated with the regression model). These discrepancies arise from differences in each statistical approach. In particular, the regression model takes all data together to estimate the slope in LPI. In contrary, the chain method is a two-step analysis in which each yearly species trend is estimated and then averaged across species. Moreover, the chain method requires interpolation between time series. The final estimates, error, degree of freedom associated with each technique are therefore different. Overall, depending on the quality of the data available the chain method (with GAMs or linear techniques) or the regression model should be preferred. The former can handle discontinuous time-series while the regression is more robust and may better describe continuous time-series data.

Change in species abundance can be positively or negatively influenced by human pressures depending on the area and species considered, which highlights the importance of considering area effects, and shows that while aggregated broad scale indices are useful, developing regional indices is important in understanding direct causal links. In particular, the increase in population size of generalist species such as herons and gulls is not necessarily synonymous of a better state of biodiversity for wetlands. Therefore, an increase in an aggregated index of population change could only reflect that some species have been able to take advantage of eutrophized, and hence more productive, degraded wetlands. In this study, we were not able to relate observed trends to ecological mechanisms. We suggest that accounting for more explicit ecological differences between species should be useful to complement biodiversity assessment based on aggregated indices of population trends. For instance, using a species-specific measure of ecological specialization should be useful to discriminate among losers from winners in current global changes (Devictor and Robert, 2009). Similarly, a community temperature index (Devictor et al., 2008) could highlight whether population abundances have been adjusted in response to temperature increase. More generally, understanding the mechanisms linking change in population abundance and the driving forces of biodiversity loss (Mace et al., 2005; Soberón and Peterson, 2009) will enable a more advanced appreciation of the causes of biodiversity change, and therefore allow a more proactive approach to conservation.

5. Conclusions

While our results suggest that at first sight, the index based on 311 species monitored in Mediterranean wetlands of different countries is positive, they also demonstrate that this general result depends on the area considered. Our analysis of index robustness suggests that extreme combination leading to negative or positive trends of LPI obtained with the chain method can arise from the removal of only few species (i.e. those with more negative or positive trends). However, if removed randomly, the average trend in LPI is more or less preserved with the chain method after the removal of up to 50% of species and is unaffected if estimated with linear regression. It can therefore be considered to be broadly robust to the species selection.

The 2010 target framework has enabled scientists to draw essential insights about available indicators, their strength and caveats, and about which information on biodiversity is still missing (e.g. Balmford et al., 2005b; Butchart et al., 2010; Mace and Baillie, 2007). Simple yet robust population monitoring can provide information that plays an important role in assessing the success of conservation policies at large temporal and spatial scales (Donald et al., 2007). But we also showed that major limitations of taking any trend in a particular indicator as a good surrogate for the state of biodiversity in a given region should be emphasized.

The species, time- and scale-dependence in trends of biodiversity have critical implications on what messages are given to the general public and politicians (Weber et al., 2004). The expansion of urban habitats and agricultural intensification, together with the disappearance of natural habitats (e.g., riverine woodlands, alluvial meadows, saline grassland, and temporary marshes) are still major causes of species decline in Mediterranean wetlands (Dudgeon et al., 2006; Eglington et al., 2008). While biodiversity losses are largely documented across the globe, assessing conservation progress can be difficult because species gains are also frequently observed at local scales (Sax and Gaines, 2003). Our results suggest that within biogeographical regions, the opposite can also be true: the positive trend of biodiversity can hide local and specific troubles. Therefore, caution is needed when assessing global trends in biodiversity from a selected list of species and spatial robustness as well as the influence of species-specific trends should be tested.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.biocon.2010.10.030.

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