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## Global changes in coastal wetlands of importance for non-breeding shorebirds



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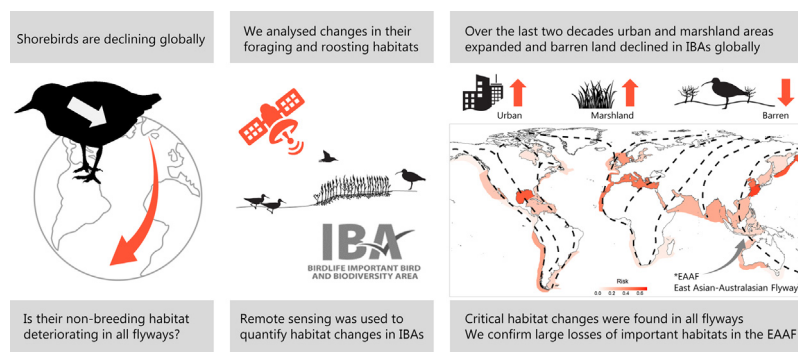
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### HIGHLIGHTS

- Shorebird declines are occurring worldwide but the causes are not fully understood.
- We analyzed habitat changes of 907 coastal wetlands important for shorebirds.
- There was an expansion of marshland and urban areas, and a decline of barren land.
- Critical habitat changes were frequent in all flyways over the last two decades.
- We confirmed the large habitat losses in the East Asian - Australasian Flyway.

### GRAPHICAL ABSTRACT



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### ABSTRACT

Shorebird declines are occurring worldwide but the causes are not fully understood. Recent literature suggests that the deterioration of habitat quality at their non-breeding areas, mostly located in temperate and tropical coastal wetlands, might be a major contributing factor. However, most studies carried out so far tend to be restricted to a few regions. Remote sensing can help correct such geographical bias on knowledge by providing a standardized approach on how shorebird habitats have been changing over the last few decades at a global scale. Here we analyzed time series of remote sensing classifications of tidal flats and land cover to quantify worldwide habitat changes in coastal Important Bird and Biodiversity Areas (IBAs) relevant for non-breeding shorebirds over the last two decades. Globally, supratidal areas (used as roosting habitat) have changed more significantly than tidal flats (used as feeding habitat). Yet, we found striking losses of tidal flats in IBAs distributed in several regions of the East Asian - Australasian Flyway. At supratidal areas, there was a general expansion of marshland, grassland and urban areas, contrasting with a decline of barren land, woodland and cropland. The expansion of marshland occurred in IBAs of most regions of the world. Urban areas also expanded consistently in supratidal areas within the most populated regions of the world. The loss of barren land is particularly concerning as it may translate into a loss of high-quality roosts and it was highly frequent in IBAs of all migratory flyways. Overall, our results confirm the large losses of shorebird habitat in the East Asian - Australasian Flyway reported in the literature, and highlight unreported generalized changes in supratidal habitats, such as the expansion of marshland and the loss of barren land, that may have negative implications for shorebirds, deserving further research and consideration in conservation programs.

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## 1. Introduction

Over a half of the shorebird populations with known trends are currently declining (BirdLife International, 2021a). The declines are widespread, extending to all migratory flyways, and have a higher incidence in migratory species (BirdLife International, 2021a). Most of these species breed in arctic and subarctic regions during the boreal summer and migrate to temperate and tropical areas during the non-breeding season (Colwell, 2010). The magnitude and taxonomic spread of shorebird declines and the evidence that temperatures are rising at a faster rate in the Arctic than in the other areas of the globe (IPCC, 2013) suggests that global warming may be an ultimate cause for the declines. However, despite the concerning ecological changes occurring at the breeding grounds (e.g. changes in breeding habitat and prey-predator dynamics; Aharon-Rotman et al., 2015; Wauchope et al., 2017), there is no evidence, yet, that the higher temperatures in the Arctic are affecting negatively the survival and recruitment of shorebirds (Aharon-Rotman et al., 2015; Weiser et al., 2018a; Weiser et al., 2018b).

The vast majority of migratory and many non-migratory shorebird species use coastal wetlands during the non-breeding season (Colwell, 2010). Here, they mostly feed on benthic macroinvertebrates in intertidal areas during the low tide and roost on open supratidal areas during high tide (Colwell, 2010). These coastal areas are also experiencing global changes with the potential to drive generalized population declines among shorebirds (Pearce-Higgins et al., 2017). Sea level rise is a global driver of transformations in coastal areas, contributing to the erosion and submergence of feeding areas of shorebirds (Galbraith et al., 2002; Nicholls and Cazenave, 2010), and to the spread of salt-tolerant vegetation into their roosting areas (Jackson et al., 2021; Straw and Saintilan, 2006). These effects are exacerbated in areas with ongoing ground subsidence and those where coastal defenses prevent the landward migration of tidal habitats (Nicholls and Cazenave, 2010). The human use of coastal areas and their surroundings has also expanded globally over the last few decades, impacting shorebirds through the claim of intertidal and supratidal areas, disturbance, pollution, hunting and competition for food resources (e.g. shellfishing; Pearce-Higgins et al., 2017; Sutherland et al., 2012).

While there is abundant literature on threats to shorebirds in coastal wetlands, the knowledge that has been produced is highly clustered in a few well-studied locations (Pearce-Higgins et al., 2017; Stroud et al., 2006). However, a global view on critical habitat changes for shorebirds is most needed, in order to identify conservation priorities and better plan the allocation of resources, boosting the development of large-scale conservation initiatives and policies that are critical to migratory species (Amano et al., 2018).

Remote sensing has assumed a key role in the monitoring of coastal wetlands over the last decades (McCarthy et al., 2017), and offers a unique opportunity to develop systematic estimates of environmental changes that are experienced by shorebirds, particularly in remote and understudied areas. Satellite imagery have been collected globally for more than five decades, covering a period when climate change effects and human impacts on natural resources escalated. In addition, the recent advent of cloud computing platforms for geospatial analysis (such as Google Earth Engine, GEE; Gorelick et al., 2017) has enabled the processing of these large imagery collections with minimal computational requirements. Recent studies have used these resources to map tidal flats (Murray et al., 2022a; Murray et al., 2019) and land cover (Sulla-Menashe and Friedl, 2018) at global scale. Here, we used publicly available products resulting from these studies to quantify habitat changes in Important Bird and Biodiversity Areas (IBAs) relevant for shorebirds over the last two decades. IBAs are priority conservation areas identified worldwide based on internationally agreed criteria of global importance for bird populations, although many are not legally protected (Donald et al., 2019). We specifically analyzed trends and quantified the gains and losses of tidal flats and land cover classes of adjacent supratidal areas in IBAs of importance for non-breeding shorebirds. With this approach, we aimed to provide a global perspective of how and where non-breeding coastal habitats for shorebirds are changing, which

ultimately can improve our understanding of the factors contributing for shorebird declines.

## 2. Methods

### 2.1. Selection of important areas for non-breeding shorebirds

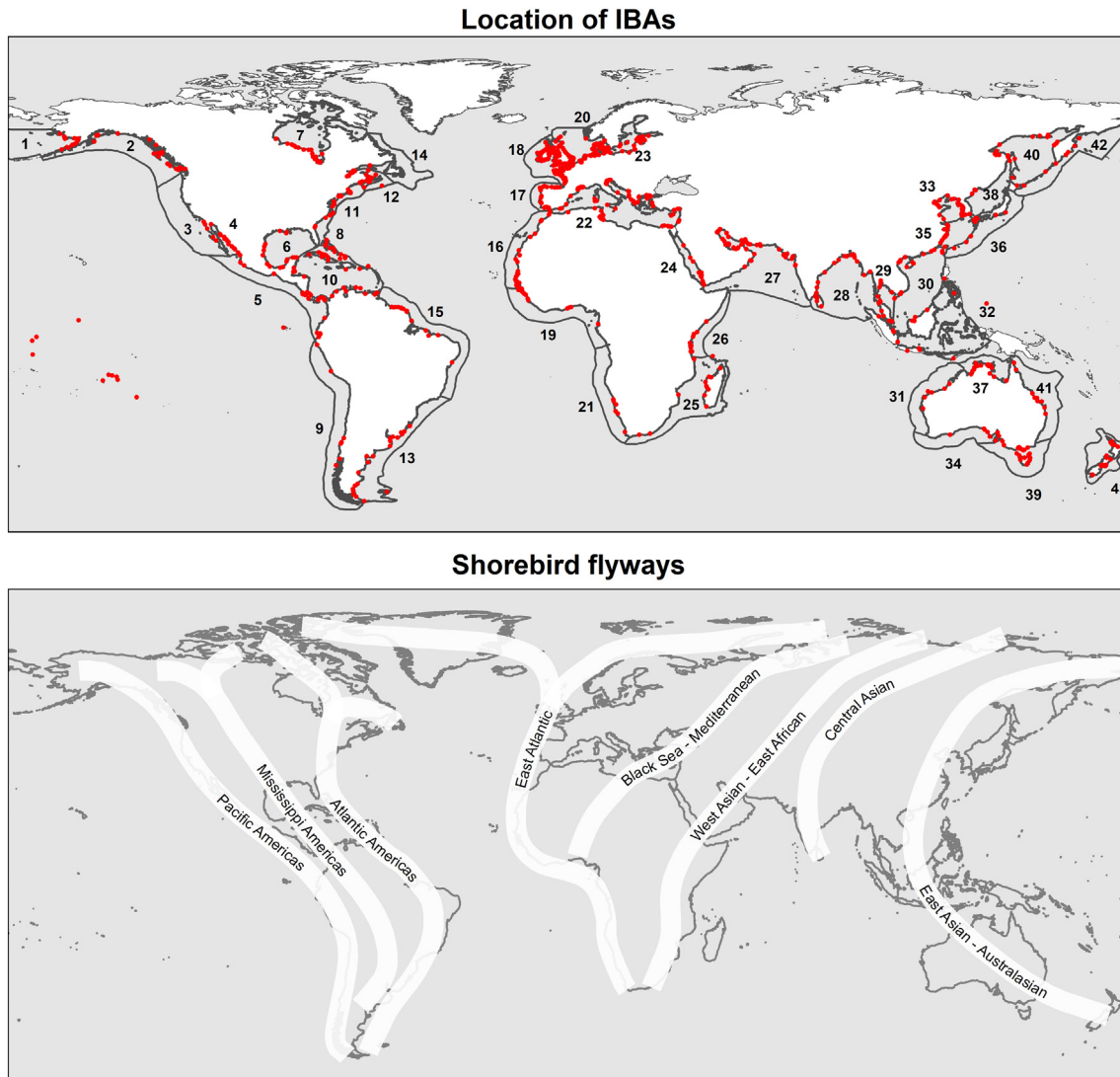
To produce a global representation of areas important for non-breeding shorebirds, we selected all IBAs with >200 ha of tidal flats (tidal flat data detailed below) hosting important populations of shorebirds of the families Charadriidae, Dromadidae, Haematopodidae, Pluvianellidae, Recurvirostridae or Scolopacidae, or congregations of waterbirds (following the IBA criteria; Donald et al., 2019). IBA boundaries and classification information were provided by BirdLife International (BirdLife International, 2021b). IBAs located at latitudes higher than 60° North and 60° South were excluded, because the distribution of tidal flats is not mapped in those areas (see below). It should be noted that while these criteria targeted non-breeding shorebirds, the selected IBAs may also be important for non-migratory species that breed in coastal habitats (del Hoyo et al., 2018). Our selection comprised 907 IBAs distributed at coastal areas around the world (Fig. 1).

### 2.2. Habitat data

Changes of intertidal and supratidal habitats for non-breeding shorebirds over the last two decades were inferred from data of two datasets: (1) the Global Tidal Flat dataset (version 1.2, Murray et al., 2022a) providing the global distribution of tidal flats in periods of three years from 1999 to 2019 at 30 m resolution; (2) the MODIS Land Cover MCD12Q1.006 (Friedl and Sulla-Menashe, 2019) providing a global distribution of land cover each year from 2001 to 2020 at 500 m resolution. Both datasets derived from the classification of satellite imagery and are publicly available (the MODIS Land Cover is available from GEE (<https://earthengine.google.com>) and the Global Tidal Flat dataset is available at doi: <https://doi.org/10.6084/m9.figshare.c.5884598.v1>).

The Global Tidal Flat dataset was produced with a machine-learning classification model applied to Landsat images that intersected a 1-km buffer of the coastline located between 60° North and 60° South (Murray et al., 2022a; Murray et al., 2019). The model was trained with information representative of three cover classes: “tidal flat”, “permanent water” or “terrestrial”, and iterated over stacks of all coastal Landsat images acquired in each period of three years. The use of a large number of images in each classification assured that all tide stages were covered and enabled the model to discriminate the extent of tidal flats. The model was designed to identify surfaces with regular tidal inundation, including mudflats, sand flats, and rock-platforms, but excluding vegetation-dominated intertidal ecosystems, such as mangroves and marshes (Murray et al., 2022a; Murray et al., 2019). However, other non-vegetated coastal areas that undergo wetting and drying regimes, such as the margins of salt pans, aquaculture ponds and coastal lagoons, may also have been classified as tidal flats (Murray et al., 2022a). As these habitats are often used by non-breeding shorebirds as feeding grounds, this misclassification did not compromise our aims. Nevertheless, we highlight the broad definition of tidal flats used in this study. Tidal flat misclassifications can also occur in areas of highly turbid water, noticed as small scattered patches with blurry edges at river mouths (Murray et al., 2022a). We did a systematic inspection of the selected IBAs and found this problem only in two IBAs overlapping with large estuaries in the Amazon region (Connétable – Marine, French Guiana; Goiabal/Piratuba, Brazil), and excluded them from the tidal flat change analysis. The classification accuracy of this dataset is 86.1% (Murray et al., 2022a).

The MODIS Land Cover dataset was produced from MODIS Terra and Aqua imagery (Sulla-Menashe et al., 2019). It was developed from a hierarchical supervised classification model where classes included in each level of the hierarchy reflect structured distinctions between land cover properties (Sulla-Menashe et al., 2019). The model also incorporated a state-



**Fig. 1.** Location of IBAs relevant for non-breeding shorebirds (red dots in the top panel) and major migratory flyways (following Boere and Stroud, 2006). IBAs were considered relevant for non-breeding shorebirds if including >200 ha of tidal flats and holding relevant populations of shorebirds or congregations of waterbirds according to BirdLife International criteria (Donald et al., 2019). Polygons with numbers in top panel represent Large Marine Ecosystems (LMEs), which are sections of ocean space along the Earth's continental margins individualized by unique undersea topography, current dynamics, marine productivity, and food chain interactions (Sherman, 1991). West Pacific Flyway was not represented for having a large overlap with the East Asian - Australasian Flyway and few exclusive IBAs. LME identification: 1 - East Bering Sea, 2 - Gulf of Alaska, 3 - California Current, 4 - Gulf of California, 5 - Pacific Central-American Coastal, 6 - Gulf of Mexico, 7 - Hudson Bay Complex, 8 - Southeast U.S. Continental Shelf, 9 - Humboldt Current, 10 - Caribbean Sea, 11 - Northeast U.S. Continental Shelf, 12 - Scotian Shelf, 13 - Patagonian Shelf and South Brazil Shelf, 14 - Labrador - Newfoundland, 15 - North Brazil Shelf and East Brazil Shelf, 16 - Canary Current, 17 - Iberian Coastal, 18 - Celtic-Biscay Shelf, 19 - Guinea Current, 20 - North Sea, 21 - Benguela Current, 22 - Mediterranean Sea, 23 - Baltic Sea, 24 - Red Sea, 25 - Agulhas Current, 26 - Somali Coastal Current, 27 - Arabian Sea, 28 - Bay of Bengal, 29 - Gulf of Thailand, 30 - South China Sea, 31 - West Central Australian Shelf and Northwest Australian Shelf, 32 - Indonesian Sea and Sulu-Celebes Sea, 33 - Yellow Sea, 34 - South West Australian Shelf, 35 - East China Sea, 36 - Kuroshio Current, 37 - North Australian Shelf, 38 - Sea of Japan, 39 - Southeast Australian Shelf, 40 - Sea of Okhotsk, 41 - East Central Australian Shelf and Northeast Australian Shelf, 42 - West Bering Sea and Oyashio Current, 43 - New Zealand Shelf. In 13, 15, 31, 32, 41, 42 two LMEs were aggregated to contain at least five IBAs.

space multitemporal modeling framework to reduce spurious land cover changes introduced by classification uncertainty in individual years (Sulla-Menashe et al., 2019). The dataset provides eight classification legends (i.e. sets of land cover categories) commonly used in land cover analysis (Sulla-Menashe and Friedl, 2018). We used the International Geosphere-Biosphere Programme (IGBP) legend (Sulla-Menashe and Friedl, 2018), which provides the best discrimination of the supratidal habitats potentially relevant for non-breeding shorebirds (see below). However, we adopted a simplified version of this classification by combining all classes with woody plants into two generic classes of woodland, and by merging the two classes of cropland. We also did not analyze snow cover, as its presence in IBAs was negligible (occurred in <2% of IBAs

with a mean cover of 0.4%), or permanent water bodies, as in most cases it was not possible to determine whether they were connected to the sea, which was masked for the purposes of the supratidal change analysis.

We considered the following seven classes of land cover:

- 1) Dense woodland - forests and scrublands (1-2 m height) with >60% of cover. Corresponding to 1–6 IGBP classes.
- 2) Sparse woodland - areas with trees and scrubs (1-2 m height) covering between 10 and 60%. Corresponding to 7–9 IGBP classes.
- 3) Grasslands - areas dominated by herbaceous annuals.
- 4) Marshland - inundated areas with vegetation. This class was originally named “Permanent Wetlands”, but in supratidal areas it corresponds



to a broad definition of Marshland, which may include low density mangrove in tropical areas. We further verified that in our study areas ca. 80% of the supratidal area covered by this class was located at <500 m from tidal flats.

- 5) Croplands - areas with >40% of cultivated cropland. Corresponding to 12 and 14 IGBP classes.
- 6) Urban - areas of human occupation with at least 30% impervious surface.
- 7) Barren land - non-vegetated areas of sand, rock and soil covering >60%.

The accuracy of the MODIS Land Cover model applied to the IGBP legend is 67%, with omission and commission errors being the highest among classes with woody plants (forests, shrublands and savannas; Sulla-Menashe et al., 2019), which justified the simplification of those classes in our analysis. Classification accuracy was also relatively low in areas with small agriculture fields surrounded by natural vegetation, which are particularly common in the tropics (Sulla-Menashe et al., 2019). Despite the identification of urban areas was relatively accurate (Sulla-Menashe et al., 2019), during a systematic inspection of the study sites we found that areas of barren land used as dirt roads, parking places and trails were often classified as urban. Although these areas did not present an impervious surface, they still represent a level of human occupation that may affect shorebirds negatively. We also emphasize that despite the improvements of this model to reduce spurious land cover changes, the classification results are still affected by a significant stochastic variability (Sulla-Menashe et al., 2019). To overcome this problem, as well as that of the systematic misclassifications described above for both datasets, we based our conclusions on trends indicating significant changes over the entire time series relying on conservative regression analysis (detailed below).

### 2.3. Data processing and analysis

Data used to analyze changes over time in tidal flats and supratidal land cover of IBAs were extracted from the Global Tidal Flat and MODIS Land Cover datasets in GEE. We quantified the tidal flat coverage as the number of  $30 \times 30$  m pixels classified as tidal flats that overlapped with the IBA polygons. Supratidal areas potentially relevant for non-breeding shorebirds were defined as a 2-km supratidal zone bordering the limit of tidal flats (i.e. the upper limit of the combined tidal flat distributions from 1999 to 2019). Non-breeding shorebirds typically roost in supratidal areas relatively close to feeding areas, such as saltmarshes, beaches, sand spits, salt plains, dunes, salt pans and mangroves (Colwell, 2010). The 2-km distance of the supratidal zone covered the range between the tidal area limit and the location of the majority of shorebird roosts identified in several studies (Bakker et al., 2021; Colwell et al., 2003; Conklin et al., 2008; Dias et al., 2006;

Handel and Gill, 1992; Rogers, 2003; Rogers et al., 2006). The 2-km supratidal zone polygons were created in GEE for IBAs and then used to extract land cover classes from the MODIS Land Cover dataset. Despite the 500 m resolution of this dataset, we extracted the data using a 100 m resolution grid in order to improve the representation of the areas near the limits of the supratidal zone polygons. IBAs where the supratidal zone covered <50 ha were not included in the supratidal change analysis (44 out of the 907 IBAs selected).

We estimated IBAs specific trends of tidal flats and supratidal land cover classes by fitting robust linear regressions using coverage (number of pixels) as response variable and the year as predictor. These regressions were fitted with Gaussian distributions using the function `rlm` of the MASS R-package (Venables and Ripley, 2002) and the function `f.robfest` of the `sfsmisc` R-package (Maechler et al., 2021) to estimate *p*-values. Robust regression iteratively optimizes weights assigned to each data point to reduce influence of outliers in the calculation of model coefficients, thus prevented that potential classification errors affected our results to any significant extent (Venables and Ripley, 2002). This analysis was not conducted for cases where the land cover classes occupied <1% of supratidal areas. We also estimated global trends for tidal flats and supratidal land cover classes by combining data from all IBAs, for which we used Generalized Linear Mixed Models (GLMM). For the supratidal areas, we modelled the proportion of supratidal area covered by each land cover class in each IBA, each year. For the intertidal areas, we modelled the proportion between tidal flat coverage each year and the accumulated tidal flat coverage from 1999 to 2019. GLMMs included year as predictor and IBA identifier as random intercept factor, and were fitted with a Quasi-Binomial distribution to control for overdispersion (Zuur et al., 2009). For this analysis we used the `glmmPQL` function of the MASS R-package (Venables and Ripley, 2002). Marginal and conditional  $R^2$  were estimated using the function `r.squaredGLMM` of the R-package `MuMIn` (Bartoń, 2019). Model overdispersion was tested with the function `testDispersion` of the `DHARMA` R-package (Hartig and Lohse, 2021).

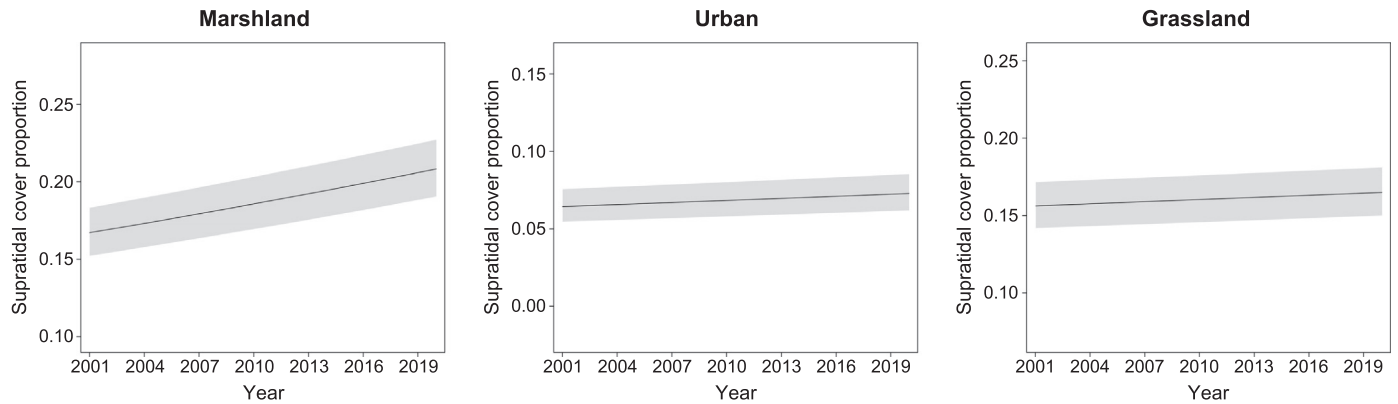
To better visualize the distribution of changes across the globe, trends of individual IBAs were grouped according to Large Marine Ecosystems (LMEs) (Fig. 1). LMEs are sections of ocean space along the continental margins individualized by unique undersea topography, current dynamics, marine productivity, and food chain interactions (Sherman, 1991). Therefore, the consistency of trends in sets of IBAs within these regions were assumed to represent regional patterns of environmental change. In some cases, we combined contiguous LMEs in order to contain at least five IBAs (cases identified in Fig. 1). Regional trends were not examined for LMEs with less than five IBA-specific trends.

To understand on how changes in intertidal and supratidal habitats may be impacting non-breeding shorebirds, we overlapped shorebird species

**Table 1**

Summary of GLMMs modeling global trends of tidal flats and supratidal land cover classes. For supratidal areas, we used the proportion of supratidal area covered by each land cover class as model response variable, and for tidal we used the proportion between tidal flat coverage each year and the accumulated tidal flat coverage from 1999 to 2019. Models included IBA identifier as random intercept factor and were fitted with a Quasi-Binomial distribution to control for overdispersion. Significant relationships are plotted in Figs. 2 and 3. Marginal and conditional  $R^2$  were calculated with the function `r.squaredGLMM` of the `MuMIn` R-package (Bartoń, 2019).

Model	Parameter	Estimate	SE	DF	<i>t</i>	P-value	$R^2$ cond./marg.
Intertidal	Intercept	-2.348	2.382	5429	-0.99	0.324	0.12/1.6 × 10 <sup>-5</sup>
	Year	0.001	0.001	5429	1.25	0.212	
Marshland	Intercept	-30.029	0.578	14,002	-51.94	<0.001	0.32/8.9 × 10 <sup>-4</sup>
	Year	0.014	2.9 × 10 <sup>-4</sup>	14,002	49.64	<0.001	
Sparse woodland	Intercept	23.546	0.703	13,546	33.47	<0.001	0.29/7.4 × 10 <sup>-4</sup>
	Year	-0.012	3.5 × 10 <sup>-4</sup>	13,546	-35.30	<0.001	
Grassland	Intercept	-8.510	0.830	12,843	-10.25	<0.001	0.28/5.0 × 10 <sup>-5</sup>
	Year	0.003	4.1 × 10 <sup>-4</sup>	12,843	8.28	<0.001	
Barren land	Intercept	17.334	1.328	5205	13.05	<0.001	0.65/2.4 × 10 <sup>-4</sup>
	Year	-0.009	6.6 × 10 <sup>-4</sup>	5205	-14.20	<0.001	
Dense woodland	Intercept	4.776	1.066	6041	4.48	<0.01	0.31/4.2 × 10 <sup>-5</sup>
	Year	-0.003	5.3 × 10 <sup>-4</sup>	6041	-6.38	<0.001	
Cropland	Intercept	5.088	1.262	5870	4.03	<0.001	0.30/4.7 × 10 <sup>-5</sup>
	Year	-0.003	6.3 × 10 <sup>-4</sup>	5870	-5.39	<0.001	
Urban	Intercept	-16.563	0.489	4483	-33.89	<0.001	0.16/1.4 × 10 <sup>-4</sup>
	Year	0.007	2.4 × 10 <sup>-4</sup>	4483	29.02	<0.001	



**Fig. 2.** Global increasing trends of supratidal areas covered by marshland, urban uses and grassland in IBAs relevant for non-breeding shorebirds. Trends were obtained from GLMMs using supratidal cover proportion in each IBA as response variable, year as predictor and IBA identifier as random factor (see model details in Table 1). Shading represents 95% confidence intervals.

distributions (BirdLife International and Handbook of the Birds of the World, 2020) with critical habitat changes (hereafter named risk factors), namely: (1) decline of tidal flat coverage, which represents direct loss of potential feeding areas (Zhang et al., 2018); (2) decline of barren land coverage in supratidal areas, representing the potential loss of high-quality roosting areas (Colwell, 2010); (3) urban development in supratidal areas; and (4) expansion of cropland in supratidal areas. The last two factors may cause the direct loss of roosts and/or elevate disturbance in roosts and feeding areas (Kirby et al., 1993; Pfister et al., 1992). Shorebird distributions were summarized as species richness, the number of species with declining trends (according to BirdLife International, 2021a) and the number of Threatened or Near-Threatened species (following the IUCN Red List Categories; IUCN, 2021).

### 3. Results

IBAs important for non-breeding shorebirds are distributed worldwide in coastal areas of temperate and tropical zones (Fig. 1).

Globally, the extent of tidal flats within IBAs did not change significantly over the last two decades (Table 1). In contrast, land cover of supratidal areas changed considerably. We found significant increases in the proportion of supratidal area covered by marshland, grassland and urban areas (Fig. 2, Table 1), and decreases in the proportion of barren land, woodland and cropland (Fig. 3, Table 1).

The trends in these classes showed a considerable heterogeneity across the globe (Figs. 4 and 5). The most consistent geographical patterns were found for marshland, urban areas and barren land. Marshland showed increasing trends in most regions of the world (Fig. 4), although with relatively low gains from 2001 to 2020 (Fig. 4). Urban areas also showed predominantly increasing trends, although for many IBAs located in remote regions trends of urban areas were not evaluated due to their low coverage in supratidal areas (Fig. 4). Gains of urban areas were generally very low, except in IBAs of East China Sea where ca. of 6% of supratidal areas became occupied by urban areas between 2001 and 2020 (Fig. 4). Barren land decreasing trends were also widespread, but as for urban areas there were some regions where trends could not be evaluated (Fig. 4). As in the two earlier cases, trends of barren land had little expression in terms of area gains or losses.

Sparse woodland and grassland trends in supratidal areas also showed some spatial consistency (Fig. 5). Grassland showed mostly increasing trends in IBAs located at high latitudes in the northern hemisphere, South America and Africa below the equator, and decreasing trends at lower latitudes of the northern hemisphere, Indonesia and Australia (Fig. 5). A nearly opposite pattern was observed for sparse woodland (Fig. 5). Area gains and losses of grassland and sparse woodland were more significant than those observed for the remaining land cover classes, with changes higher than 5% of the supratidal area detected in several regions (Fig. 5).

Trends for dense woodland, cropland and tidal flats showed less spatial consistency (Figs. 4 and 5). However, striking losses of tidal flats were recorded along the East Asian - Australasian Flyway, with several regions showing tidal flat coverage decreases over 15% from 1999 to 2019 (Fig. 4).

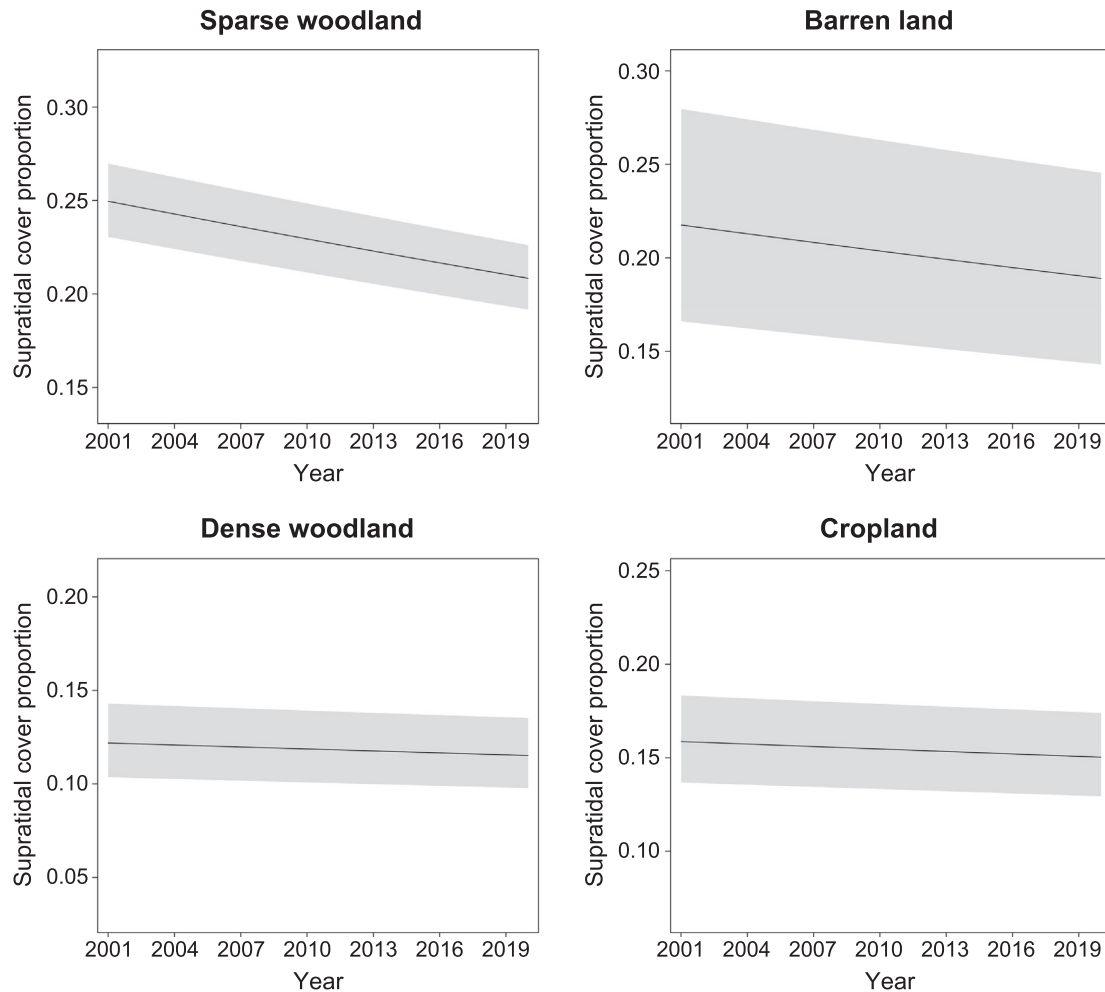
The most relevant shifts in supratidal areas over the last two decades involved grassland and sparse woodland, with the transformation of sparse woodland into grassland being greater than the reverse (Fig. 6). Marshland gains occurred mostly at areas previously occupied by sparse woodland and grassland, although the reverse transformation also occurred (Fig. 7). Barren land showed important exchanges with grassland and significant losses to marshland (Fig. 7). Dense woodland was mostly lost to sparse woodland and marshland (Fig. 7). Cropland showed relevant exchanges with grassland and sparse woodland (Fig. 7). Urban areas mostly expanded to supratidal areas previously occupied by sparse woodland and grassland (Fig. 7). Exchanges of intertidal areas occurred mostly with open water and marshland (Fig. 7). The extent of supratidal land cover classes and shifts on those classes and tidal flats for specific geographical regions are presented in supplementary materials (Tables S1, S2 and S3).

Among changes considered as risk factors for non-breeding shorebirds, the decline of barren land was the most frequent in IBAs (51% of the IBAs), followed by urban and cropland expansion (47% and 32% respectively, Fig. 8a). The decline of tidal flats occurred in a lower percentage of IBAs (11%, Fig. 8a). These factors were frequent in all migratory flyways, but a particularly high incidence was found in the Black Sea - Mediterranean Flyway, the East Asian - Australasian Flyway, the East Atlantic Flyway and the Mississippi Americas Flyway (Fig. 8a, b). When combined with shorebird species distribution, risk factors assumed particular relevance in tropical and subtropical regions of the northern hemisphere, overlapping with the distribution of more species in general but also more species with a declining population trend and with a conservation status of Threatened or Near-Threatened according to IUCN (Fig. 8d, f, h).

### 4. Discussion

Our results showed that, over the last two decades, coastal IBAs relevant for non-breeding shorebirds suffered more changes in supratidal areas than in tidal flats. There was a general expansion of marshland, grassland and urban areas, and a decline of barren land, woodland and cropland (Figs. 2 and 3), although most of these trends showed a significant regional heterogeneity (Figs. 4 and 5). Tidal flats did not show a significant general trend, but IBAs of several regions of the East Asian - Australasian Flyway showed striking tidal flat losses (Fig. 4).

Marshland presented the most consistent geographical patterns, with increasing trends in almost all regions of the world (Fig. 4). This broad land cover class represent inundated areas with vegetation (including sparse mangrove) and was mostly distributed along the edge of tidal flats in the studied IBAs (see Methods). The generalized increasing trend of



**Fig. 3.** Global decreasing trends of supratidal areas covered by woodland, barren land, and cropland in IBAs relevant for non-breeding shorebirds. Trends were obtained from GLMMs using supratidal cover proportion in each IBA as response variable, year as predictor and IBA identifier as random factor (see model details in Table 1). Shading represents 95% confidence intervals.

marshland in supratidal areas suggests a relationship with sea level rise, as the expansion of saltwater inland is typically followed by the development of tidal marshes and mangroves (Asbridge et al., 2016; Donnelly and Bertness, 2001; Raabe and Stumpf, 2016; Visschers et al., 2022). Marshland expanded mostly to areas previously covered by sparse woodland and grassland, these being the following main land cover classes of supratidal areas (Figs. 6 and 7). Yet, important losses of barren land, dense woodland and even tidal flats were due to the expansion of marshland (Fig. 7). Despite the consistent expansion trends in most regions, the area gains of marshland were never higher than 5% of the supratidal area (Fig. 4).

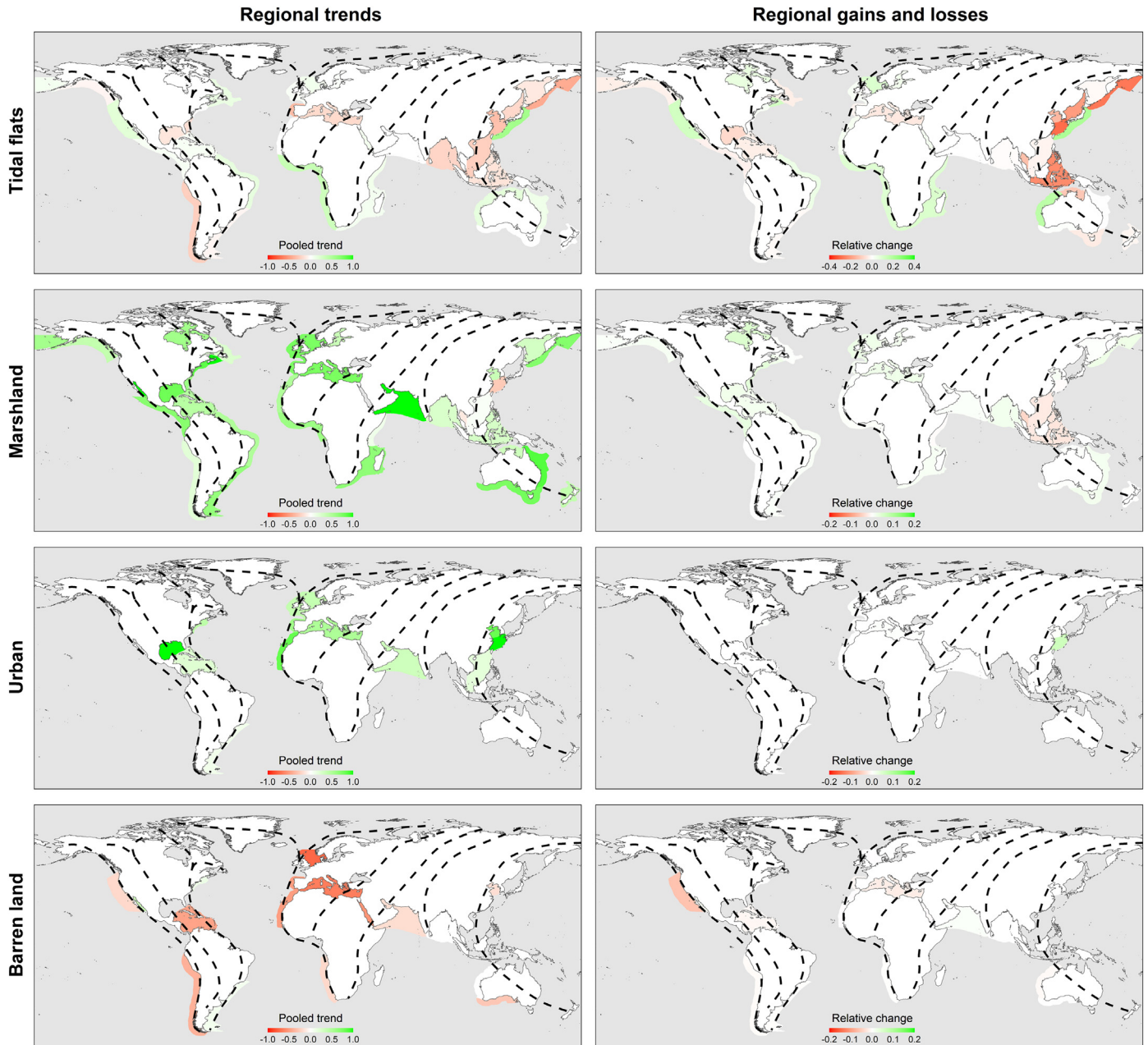
Whether or not the expansion of marshland in supratidal areas represents a threat to shorebirds is difficult to ascertain. Some species are well adapted to feed on marshlands, and even species that mainly forage in tidal flats may also feed supplementary in these habitats during the high tide (van de Kam et al., 2004). Some species also roost on saltmarshes and mangroves (Conklin et al., 2008; Johnston-Gonzalez and Abril, 2019). However, most shorebirds tend to concentrate in larger numbers in supratidal unvegetated areas at nearshore, from where they have an unobstructed view of the horizon and may react more quickly to approaching predators (Colwell, 2010). In this context, the conversion of barren land into marshland may represent a potential threat for non-breeding shorebirds, as it may translate into the loss of high-quality roosts, such as beaches, sand spits and salt plains (Colwell, 2010).

Further concerns are due to the fact that barren land has been declining in most regions where we were able to quantify its trends (Fig. 4). It must be noted, however, that roosting shorebirds may concentrate in relatively

small areas (Colwell, 2010). Thus, even considerable reductions of barren land in supratidal areas may not always limit the availability of high tide roosts for shorebirds, particularly in IBAs located in arid regions where barren land covers vast supratidal areas.

The expansion of urban areas was also widespread in IBAs, particularly those in the temperate regions of the northern hemisphere, corresponding to the most populated areas of the world (Fig. 4). The losses of supratidal areas due to urban development were generally smaller than 2% (except in IBAs of East China Sea, Fig. 4), but that does not diminish the severity of this impact. Besides the complete destruction of habitat potentially relevant for shorebirds, urban development in supratidal areas generally translate into disturbance of shorebird roosts and feeding areas by humans and domestic animals that may lead to their abandonment (Kirby et al., 1993; Pfister et al., 1992; van der Kolk et al., 2022).

Shifts between grassland and sparse woodland were the most common changes in supratidal areas, with sparse woodland being replaced by grassland more often than the reverse (Fig. 6). This resulted in a global decline of sparse woodland and the expansion of grassland (Figs. 2 and 3). These changes may have been driven by climatic factors given the distributional coherence of the regional trends (Fig. 5). In fact, the distribution of regional trends of grasslands matches to some extent the projected changes in soil moisture of IPCC (IPCC, 2013), with the regions expected to suffer severer drought showing declining trends in grassland (Fig. 5). Similarly, the expansion of grasslands was mostly observed at high latitudes of the northern hemisphere (Fig. 5) where an increase of precipitation is expected due to climate change (IPCC, 2013). We did not find a clear explanation for the

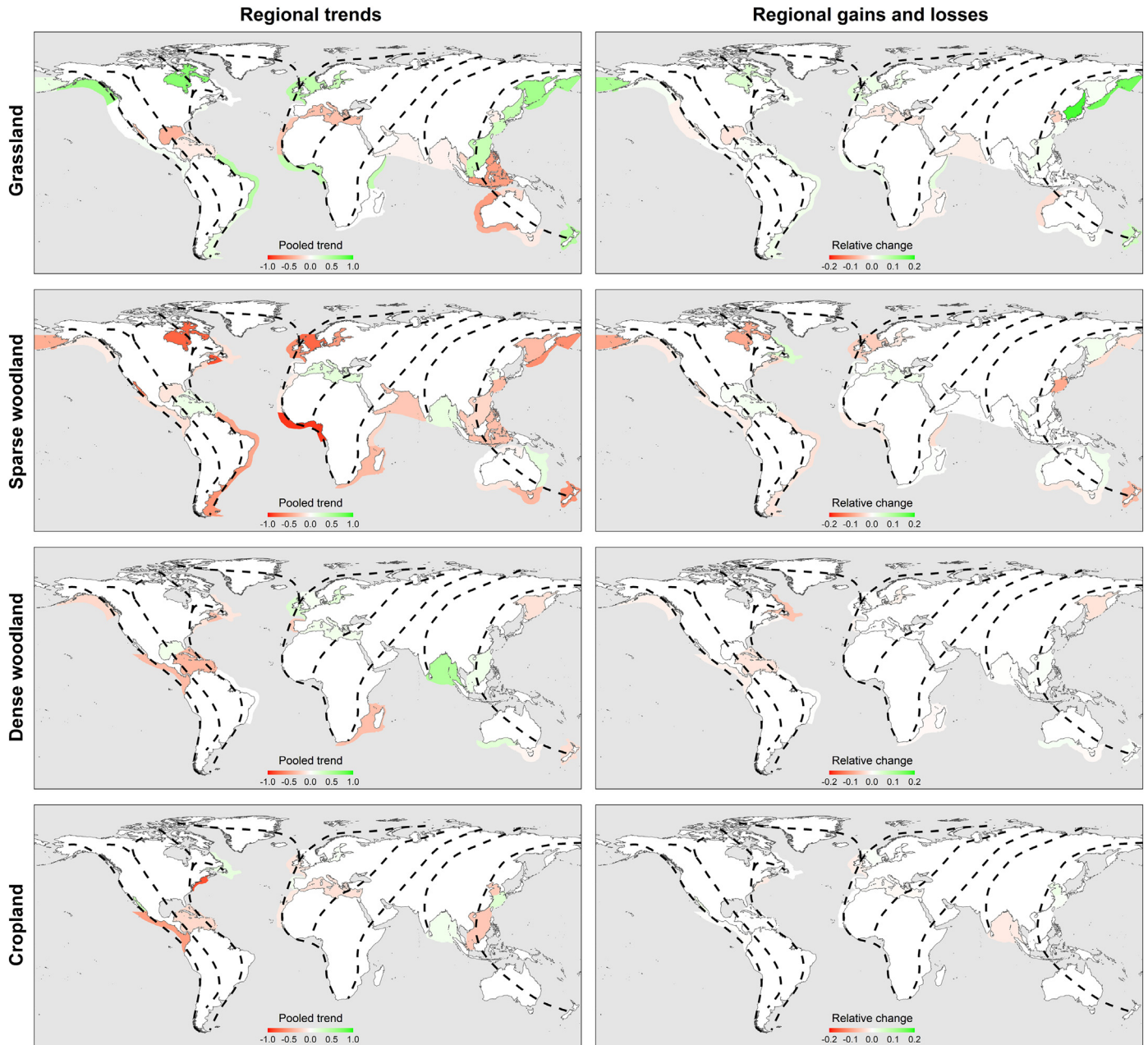


**Fig. 4.** Regional trends, gains and losses of tidal flats and supratidal areas covered by marshland, urban areas and barren land in IBAs relevant for non-breeding shorebirds. Pooled trends (shown in left panel plots) correspond to the number of IBAs with increasing trends minus the number of IBAs with decreasing trend divided by the total number of IBAs evaluated in each LME. LMEs were not colored if the number of IBAs where trends were evaluated was less than five. Trends of individual IBAs were determined through regression analysis (see [Methods](#)). Relative change (shown in right panel plots) reflects coverage losses or gains from the first to the last sampling years in each LME. For tidal flats the relative change represents the coverage difference from 1999 to 2001 to 2017–2019 divided by the tidal flat area accumulated in that period (considered the maximum area of tidal flats potentially available). For marshland, urban areas and barren land the relative change represents coverage difference from 2001 to 2020 divided by the overall supratidal area. Positive values of relative change reflect gains while negative values reflect losses. Note that the scale of relative change is different between tidal flats and supratidal land cover classes.

reduction of sparse woodland, but the inverse patterns between sparse woodland and grassland suggests that there was a relationship between these two land cover classes ([Fig. 5](#)). These patterns may be related with the land cover classification procedure, where the increase of grass density in sparse woodland may have resulted in the classification as grassland with no relevant changes in tree and scrub cover ([Sulla-Menashe et al., 2019](#)). Nevertheless, these changes have limited significance for the conservation of non-breeding shorebirds, as both habitats are little used as feeding or roosting areas ([del Hoyo et al., 2018](#)), but some shorebirds breeding in these coastal areas may benefit from the expansion of grasslands ([del Hoyo et al., 2018](#)).

The global declines of dense woodland and cropland lacked a clear geographical pattern ([Fig. 5](#)) suggesting that the changes of these land cover classes were related to local rather than global-scale factors. In addition, these changes resulted in relatively low coverage losses in supratidal areas ([Fig. 5](#)). It is worth noting, that the declining trend of cropland in supratidal areas does not necessarily translate into benefits for non-breeding shorebirds. For instance, in East and Southeast Asia large areas of cropland have been transformed into aquaculture ([Liu et al., 2020](#); [Mialhe et al., 2016](#); [Ren et al., 2019](#)), which generally are equally poor habitats for shorebirds and may present higher disturbance and pollution ([Jackson et al., 2020](#); [Melville et al., 2016](#)), with a





**Fig. 5.** Regional trends, gains and losses of supratidal areas covered by grassland, sparse woodland, dense woodland and cropland in IBAs relevant for non-breeding shorebirds. Pooled trends (shown in left panel plots) correspond to the number of IBAs with increasing trends minus the number of IBAs with decreasing trend divided by the total number of IBAs evaluated in each LME. LMEs were not colored if the number of IBAs where trends were evaluated was less than five. Trends of individual IBAs were determined through regression analysis (see [Methods](#)). Relative change (shown in right panel plots) reflects coverage losses or gains from the first to the last sampling years in each LME (i.e. coverage difference from 2001 to 2020 divided by the overall supratidal area). Positive values of relative change reflect gains while negative values reflect losses.

few notable exceptions (e.g. [Marquez-Ferrando et al., 2014](#); [Navedo et al., 2017](#)).

The lack of a clear global trend in the extent of tidal flats in IBAs is in line with previous studies. [Murray et al. \(2019\)](#) have demonstrated a declining trend of tidal flat extent worldwide for a longer time series (1984–2016) but such trend lost statistical significance when the time series was reduced to the period between 1999 and 2016. Furthermore, [Hill et al. \(2021\)](#) did not find a significant trajectory of tidal flats within coastal protected areas for the same period (1999–2016). We should emphasize that tidal flats are highly dynamic ([Murray et al., 2022b](#)) and such variation may have masked their general trajectory in IBAs. In addition, regional trends must be interpreted with caution, particularly in remote areas where recognized drivers of tidal flat loss, such as coastal land claim and

river damming ([Murray et al., 2019](#); [Murray et al., 2022b](#)), are absent. Nevertheless, the striking loss of tidal flats observed in the Korean Peninsula, China and Southeast Asia ([Fig. 4](#)) was most likely related to coastal development and conversion to aquaculture ([Liu et al., 2020](#); [Ma et al., 2014](#); [Mialhe et al., 2016](#); [Murray et al., 2022b](#); [Ren et al., 2019](#)). We stress the 30% loss of tidal flats in East China Sea, which was even higher than that recorded for the Yellow Sea ([Fig. 4](#)), an area where land claim had enormous impacts on non-breeding shorebirds ([Murray et al., 2014](#); [Piersma et al., 2016](#); [Studds et al., 2017](#)). In addition to tidal flat loss, IBAs of East China Sea have lost 6% of supratidal areas to urban development and this was the only region in the world where we found a pronounced declining trend of marshland ([Fig. 4](#)). This evidence suggests that land claim in East China Sea is the most recent environmental disaster for shorebirds in the



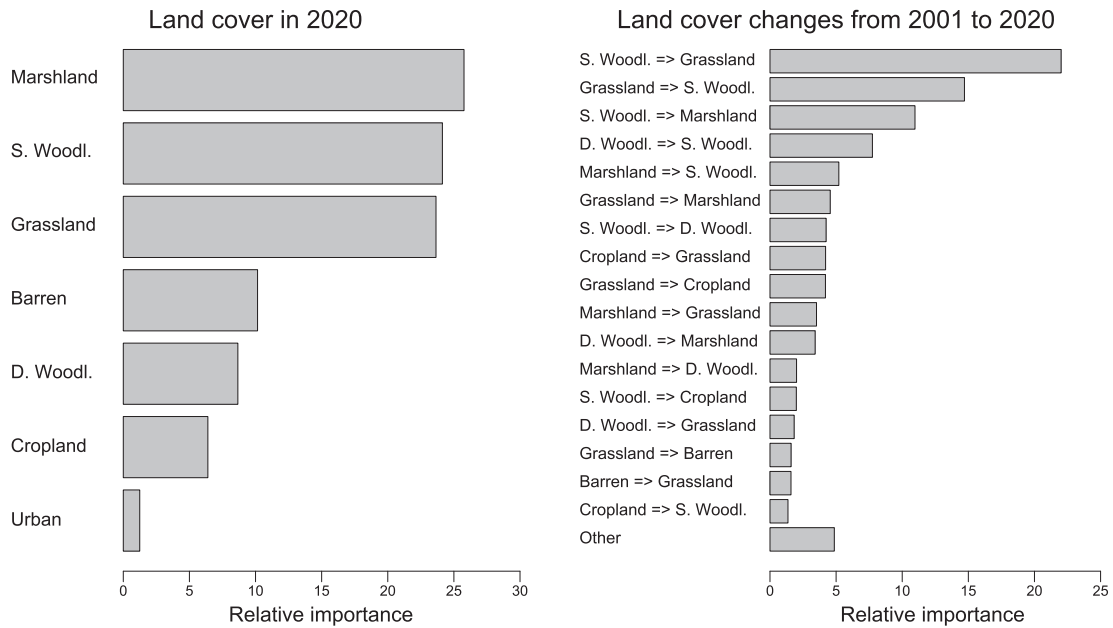


Fig. 6. Relative importance of supratidal land cover classes in 2020 and changes between 2001 and 2020 in IBAs relevant for non-breeding shorebirds. In the left panel, values represent percentage of supratidal area covered by each land cover class. In the right panel, values represent the area permuted between pairs of land cover classes relative to the overall area that changed land cover from 2001 to 2020 expressed in percentage. S. Woodl. – Sparse woodland; D. Woodl. – Dense woodland; Barren – Barren land.

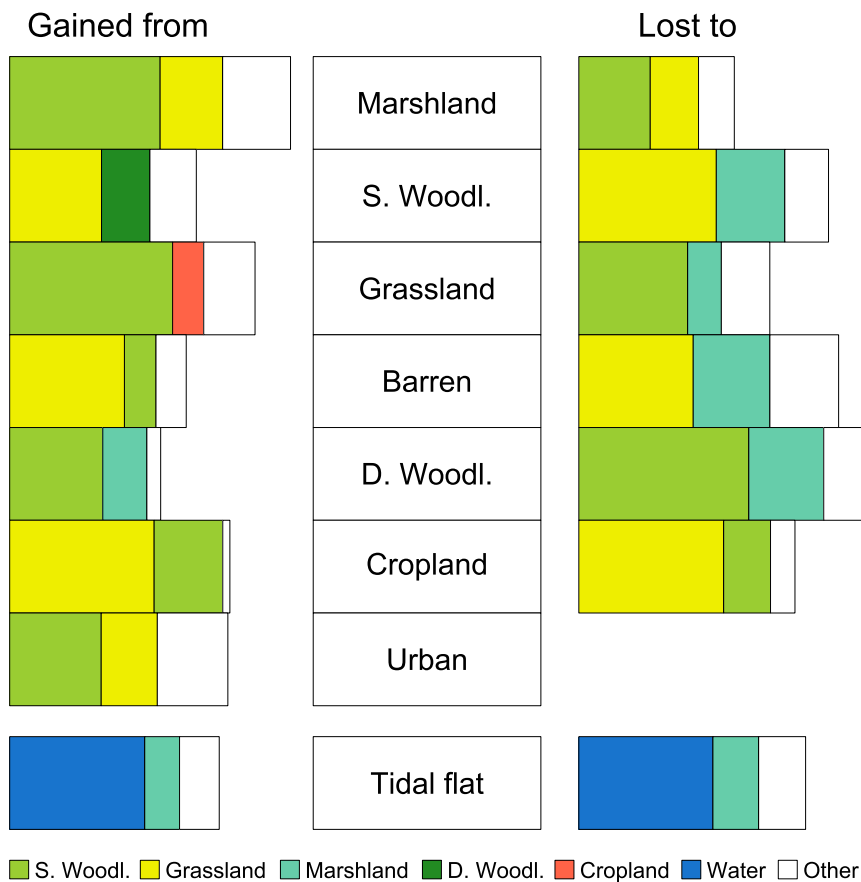
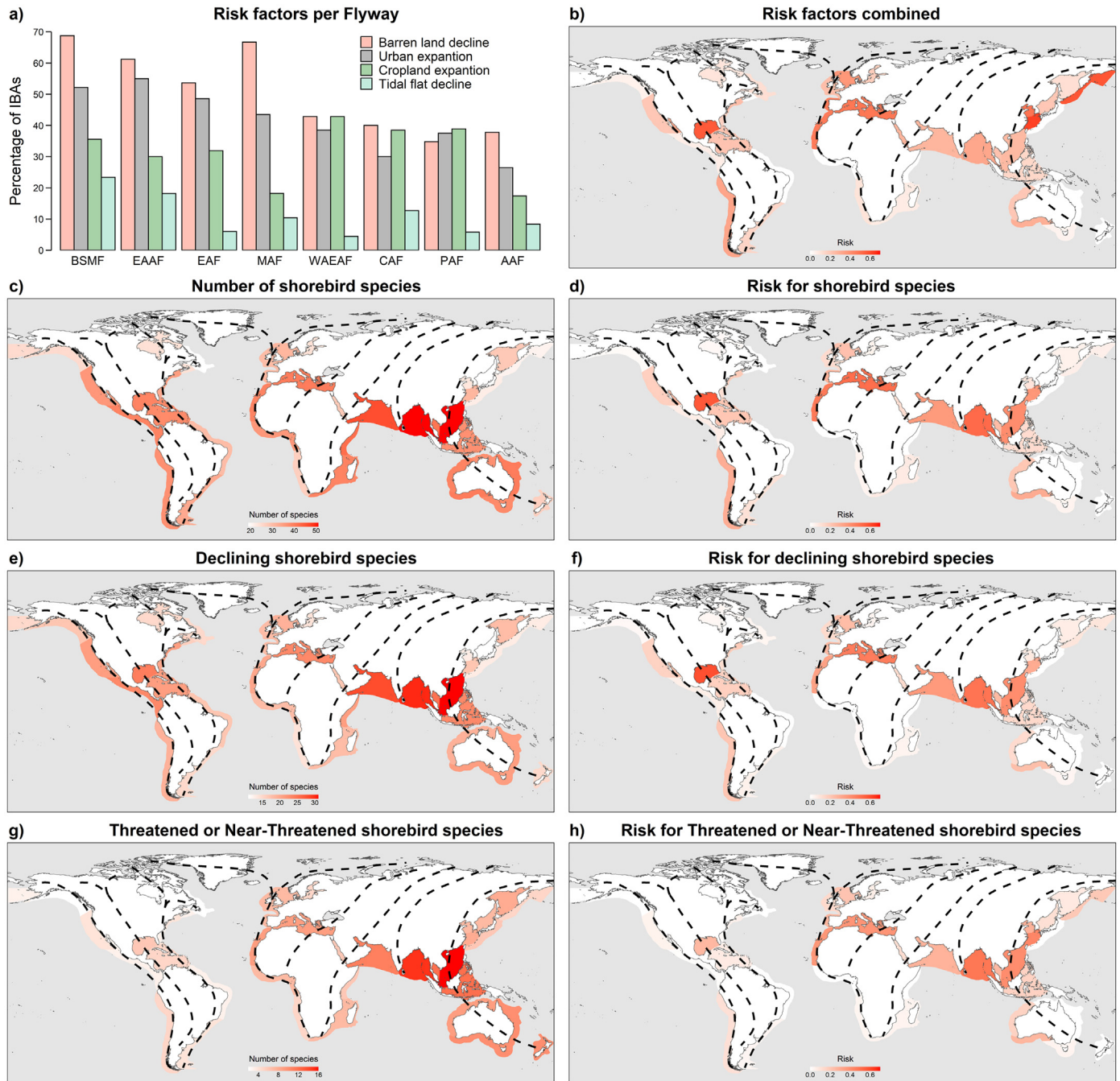


Fig. 7. Cover transitions at supratidal and intertidal areas of IBAs relevant for non-breeding shorebirds. Cover exchanges were measured between 2001 and 2020 for supratidal areas and between 1999-2001 and 2017-2019 for intertidal areas. The range of intertidal areas corresponds to the tidal flat area accumulated between 1999 and 2019. Permanent water was only considered for exchanges in intertidal areas. Bar lengths represent percentage of area permuted, where gains and losses in each land cover class sum up 100%. S. Woodl. – Sparse woodland; D. Woodl. – Dense woodland; Barren – Barren land.



**Fig. 8.** (a) Percentage of IBAs affected by risk factors in each flyway. Risk factors represent critical habitat changes for non-breeding shorebirds. BSMF - Black Sea - Mediterranean Flyway; EAAF - East Asian - Australasian Flyway; MAF - Mississippi Americas Flyway; EAF - East Atlantic Flyway; WAEAF - West Asian - East African Flyway; AAF - Atlantic Americas Flyway; CAF - Central Asian Flyway; PAF - Pacific Americas Flyway. West Pacific Flyway was not represented for having a large overlap with the East Asian - Australasian Flyway and few exclusive IBAs. (b) Global distribution of risk factors (represented over the LMEs). For this representation, we calculated the proportion of IBAs affected by each risk factor in each LMEs, and then averaged the four risk factors. (c, e, g) Distributions of shorebird species richness, declining species according to Bird-Life International, and Threatened or Near-Threatened species according to IUCN. (d, f, h) Incidence of risk factors on species richness, declining species and Threatened or Near-Threatened species. Maps were generated through the multiplication of species maps with the risk factors map, after scaling the numeric variables between 0 and 1.

East Asian - Australasian Flyway, and probably played an important role in the population declines that have been observed in recent years (Studds et al., 2017).

Among the changes considered as risk factors for non-breeding shorebirds, the decline of barren land was the most frequent in IBAs, followed by urban and cropland expansion (Fig. 8a). The decline of barren land may represent the loss of high-quality roosts, i.e. widely open areas in the proximity of feeding areas (Colwell, 2010). This factor is particularly concerning for being largely associated to the expansion of marshland, which

has been occurring worldwide (Fig. 4) and may be persistent in the future due to sea level rise (Schuerch et al., 2018). Urban and cropland expansion may result in the destruction of roosts or increase disturbance of roosts and feeding areas (Pearce-Higgins et al., 2017). The decline of tidal flats was not as frequent as the remaining risk factors in IBAs (Fig. 8a).

Altogether, risk factors were widespread, being frequent in IBAs of all migratory flyways (Fig. 8a). Still, we highlight the very high frequency of these factors in the Black Sea - Mediterranean Flyway (Fig. 8a). This flyway was already considered highly threatened by the

early 2000s due to the occurrence of a large number of declining shorebird species (Stroud et al., 2006), but it kept the record of being among the least studied flyways.

The combination of risk factors and species distribution highlight the higher vulnerability of IBAs located in tropical and subtropical regions of the northern hemisphere, some of which are also between the least studied (Fig. 8). In these regions there is a critical overlap between fast-developing economies (such as India and China) with a high concentration of shorebird species, including several species that are declining or threatened, requiring increased conservation actions.

## 5. Conclusions

This study is the first attempt to quantify global changes on relevant habitats for non-breeding coastal shorebirds based on geographical data that is currently available worldwide on the distribution of tidal flats and land cover. The results reinforce evidence reported in recent shorebird literature, such as the impacts of land claim on critical areas of the East Asian - Australasian Flyway (Murray and Fuller, 2015; Zhang and Ouyang, 2019), and also led to the identification of unreported trends of habitat change in supratidal areas that deserve further research, such as the expansion of marshland and the decline of barren land. Our study also highlighted the potential of remote sensing in identifying regions of higher risk for shorebirds, some of them clearly understudied and thus deserving further attention from the research community - such as the Black Sea - Mediterranean flyway and many tropical regions. The fast-development of remote sensing of environmental changes (boosted by the availability of new satellite sensors such as Sentinel (Desnos et al., 2014) and cloud computing (Gorelick et al., 2017)) can give a valuable contribution to filling current knowledge gaps, and offer future opportunities to investigate the environmental factors contributing to the decline of shorebirds.

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## CRedit authorship contribution statement

**Carlos D. Santos:** Conceptualization, Methodology, Formal analysis, Writing – original draft. **Teresa Catry:** Conceptualization, Writing – review & editing. **Maria P. Dias:** Conceptualization, Writing – review & editing. **José P. Granadeiro:** Conceptualization, Writing – review & editing.

## Data availability

The data used in this manuscript is publicly available. Further information is provided in the manuscript methods section.

## Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

## Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.scitotenv.2022.159707>.

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