Identification of candidate pelagic marine protected areas through a seabird seasonal-, multispecific- and extinction risk-based approach

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Abstract

With increasing pressure on the oceans from environmental change, there has been a global call for improved protection of marine ecosystems through the implementation of marine protected areas (MPAs). Here, we used species distribution modelling (SDM) of tracking data from 14 seabird species to identify key marine areas in the southwest Atlantic Ocean, valuing areas based on seabird species occurrence, seasonality and extinction risk. We also compared overlaps between the outputs generated by the SDM and layers representing important human threats (fishing intensity, ship density, plastic and oil pollution, ocean acidification), and calculated loss in conservation value using fishing and ship density as cost layers. The key marine areas were located on the southern Patagonian Shelf, overlapping extensively with areas of high fishing activity, and did not change seasonally, while seasonal areas were located off south and southeast Brazil and overlapped with areas of high plastic pollution and ocean acidification. Non-seasonal key areas were located off northeast Brazil on an area of high biodiversity, and with relatively low human impacts. We found support for the use of seasonal areas depending on the seabird assemblage used, because there was a loss in conservation value for the seasonal compared to the non-seasonal approach when using 'cost' layers. Our approach, accounting for seasonal changes in seabird assemblages and their risk of extinction, identified additional candidate areas for incorporation in the network of pelagic MPAs.

Introduction

Oceans are facing rapid and profound changes in their characteristics and structure (Halpern et al., 2008; Rockström et al., 2009; Valdés et al., 2009). Ocean changes are so widespread that only <4% of the global ocean area is considered to be experiencing very low human impacts (Halpern et al., 2015). Hence, there is a need for a major international effort to protect and maintain the functioning of marine ecosystems, through the implementation of marine protected areas (MPAs) (Rockström et al., 2009; Kachelriess et al., 2014). Despite a global agreement to designate MPAs covering 10% of the global ocean (Secretariat of the Convention on Biological Diversity, 2006), only 2.98% is currently

protected, and <1% of the global ocean comprise 'no-take' zones, where no harvesting of marine resources is allowed (Marine Conservation Institute, 2015). Furthermore, few established MPAs effectively and integrally protect their targeted species, so their value is likely to be even lower than expected (Jameson, Tupper & Ridley, 2002; Mora & Sale, 2011; Edgar *et al.*, 2014).

To truly protect the diversity of marine systems, the design of MPAs must account for its location (Jameson *et al.*, 2002; Sundblad, Bergström & Sandström, 2011), connectivity with other areas (Sundblad *et al.*, 2011), number and proportion of protected oceanographic features (e.g. upwellings, seamounts; Sundblad *et al.*, 2011; Dunn *et al.*, 2014), species occurring there and their seasonality (Hyrenbach, Forney & Dayton, 2000; Mann & Lazier, 2006), climatic variability (Jameson *et al.*, 2002), isolation from human impacts (Rife *et al.*, 2013; Edgar *et al.*, 2014), human activities (Charles & Wilson, 2009; Edgar *et al.*, 2014; Mazor *et al.*, 2014) and socio-economic use of the area (Charles & Wilson, 2009). Most MPAs are designed to improve fisheries management, that is, to manage commercial fish stocks in a more sustainable way (e.g. Pitchford, Codling & Psarra, 2007; Horta e Costa *et al.*, 2013a; Horta e Costa, Gonçalves & Gonçalves, 2013b; Stevenson, Tissot & Walsh, 2013). Few studies have accounted for other environmental factors in the design of MPAs. For instance, ship movements (Dalton, 2004; Halpern *et al.*, 2008), exposure to plastics (Lebreton, Greer & Borrero, 2012; Van Sebille, England & Froyland, 2012; Wilcox, Van Sebille & Hardesty, 2015), oil pollution (Wiese & Robertson, 2004; Moreno *et al.*, 2013; Kark *et al.*, 2015) and ocean acidification (Harvey, Gwynn-Jones & Moore, 2013) present substantial impacts to marine biodiversity, and should be considered in MPA design (Kelleher, 1999; Pomeroy *et al.*, 2005).

Wide-ranging, pelagic top predators have been widely proposed as useful tools for pointing out important areas, which can add conservation value to MPA networks (Hooker & Gerber, 2004; Sergio *et al.*, 2008; Ronconi *et al.*, 2012). Top predators integrate factors affecting seasonal variation in abundance and distribution of their lower trophic-level prey over broad areas (Montevecchi *et al.*, 2012; Tancell *et al.*, 2012; Michael, Jahncke & Hyrenbach, 2014). Seabirds, in this context, are relatively easy to sample at their breeding colonies and to track them at sea using electronic devices, providing information on foraging movements at various spatio-temporal scales (Phillips *et al.*, 2006a; Delord *et al.*, 2014; Thiebot *et al.*, 2014). Several important marine areas have been identified based on the distribution and occurrence of seabird species and proposed to integrate the current network of MPAs. The marine Important Bird Areas (mIBAs) represent one of the most recent wide-ranging international efforts to identify relevant areas for marine conservation (Lascelles *et al.*, 2012, 2016).

Here, we applied spatial distribution modelling to tracking data from 14 pelagic seabird species in the southwest Atlantic Ocean (Food and Agriculture Organization, FAO, Fishing Area 41). The use of tracking data is deliberately used to identify important areas in offshore waters since many important 'coastal' areas have been identified for MPAs or MIBAs through other approaches and investigations of coastal seabirds. We identified key areas based solely on the climatic and oceanographic variables irrespective of the distribution of threats, *a posteriori* we quantified sensitivity of species to threats and quantified the costs to human activities within those areas that are important for the birds, by imposing restrictions of activities within the areas and calculating changes in conservation value. There are evidences that the distribution of threats matches oceanographic conditions targeted by marine animals, then the approach is justified (Karpouzi, Watson & Pauly, 2007; Hatch *et al.*, 2008; Titmus & Hyrenbach, 2011; Krüger *et al.*, 2017). Our objectives were: (1) identify important areas for pelagic seabirds throughout the year; (2) quantify the sensitivity of the identified important areas for seabird assemblages to environmental pressures; (3) compare our spatially explicit results to the existing networks of proposed or established protected areas identified by international organizations, and (4) carry out a cost analysis evaluation of the implementation of seasonal versus non-seasonal protected areas. FAO Fishing Area 41 encompasses a wide range of latitudes from tropical to Antarctic waters, including a large array of habitats and oceanographic conditions, the largest continental shelf in the Southern Hemisphere and a highly productive convergence zone that supports a biodiversity-rich ecosystem, and major fisheries (Vasconcellos & Csirke, 2011). Despite several proposals to identify key areas for conservation in pelagic ecosystems, either in the form of ecologically and biologically significant areas (EBSAs) (Dunn *et al.*, 2014) or mIBAs (Lascelles *et al.*, 2012), the designated MPAs inside FAO41 are mostly restricted to coastal regions (Marine Conservation Institute, 2015, http://www.mpatlas.org/explore/). Thus, such network of MPAs does not protect marine pelagic species and systems. In this sense, our analysis prioritizes habitats used by pelagic seabird communities weighted by their seasonality and extinction risk (which may protect other pelagic species) as pelagic habitats are underrepresented in the MPAs network within FAO41. The key areas identified with this study may be seen as supporting the already proposed MPAs as add-ons, which indentifies pelagic and seasonal areas that were not considered by previous methods.

Materials and methods

Tracking data and bird assemblages

We used geolocator (global location sensor or GLS) tracking data from 14 seabird species, occurring at the southwest Atlantic Ocean during part or all of the annual cycle (Table 1). Deployment and other details are provided elsewhere (Grémillet *et al.*, 2000; Croxall *et al.*, 2005; Phillips *et al.*, 2006a; González-Solís, Croxall & Afanasyev, 2008; González-Solís *et al.*, 2009; Mackley *et al.*, 2010; Quillfeldt *et al.*, 2013; Ramírez *et al.*, 2013; Reid *et al.*, 2013; Missagia *et al.*, 2015; Krüger *et al.*, 2016, 2017). The data were modelled in three different periods: summer (October–March), winter (April–September) and year-round (January–December).

Table 1 Seabird species included in the study

Common Name	Species	Abbreviation	Size Range (cm)	Assemblage	IUCN Status	Colony	N	Years	Source
Tristan Albatross	*Diomedea dabbenena*	TA	110	Northern	CR	Gough Island	34	2004, 2005, 2006	Reid *et al.* (2013)
Wandering Albatross	*Diomedea exulans*	WA	120–135	Southern	VU	Bird Island (South Georgia)	18	2003	Mackley *et al.* (2010)
Black-browed Albatross	*Thalassarche melanophris*	BBA	80–96	Southern	NT	Bird Island (South Georgia), New Island (Falklands)	57	1996, 1997, 2002, 2003	Grémillet *et al.* (2000), Mackley *et al.* (2010)
Grey-headed Albatross	*Thalassarche chrysostoma*	GHA	70–85	Southern	EN	Bird Island (South Georgia)	35	2003, 2006	Croxall *et al.* (2005)
Northern Giant Petrel	*Macronectes halli*	NGP	80–95	Southern	LC	Bird Island (South Georgia)	25	1999, 2001	González-Solís *et al.* (2008)
Southern Giant Petrel	*Macronectes giganteus*	SGP	85–100	Southern	LC	Bird Island (South Georgia), Elephant Island (South Shetlands)	40	1999, 2001, 2011	González-Solís *et al.* (2008), Krüger *et al.* (2017)
White-chinned Petrel	*Procellaria aequinoctialis*	WCP	51–58	Southern	VU	South Georgia	11	2003, 2004	Phillips *et al.* (2006a)
Antarctic Prion	*Pachyptila desolata*	AP	25–27	Southern	LC	South Georgia	10	2009, 2010	Quillfeldt *et al.* (2013)
Deserta's Petrel	*Pterodroma deserta*	DP	35	Northern	VU	Bugio Island	24	2007, 2008, 2009, 2010	Ramírez *et al.* (2013)
Trindade Petrel	*Pterodroma arminjoniana*	TP	37–40	Northern	VU	Trindade Island	4	2013, 2014	Krüger *et al.* (2016)
Cory's Shearwater	*Calonectris borealis*	CS	46	Northern	LC	Berlenga, Azores and Canary Islands	41	2002, 2003, 2004, 2011, 2012	González-Solís *et al.* (2009), Missagia *et al.* (2015)
Great Shearwater	*Ardenna gravis*	GS	46–51	Southern	LC	Gough Island	37	2008 –2012	González-Solís
Manx Shearwater	*Puffinus puffinus*	MS	30–35	Southern	LC	Heimaey, Iceland	10	2006, 2007	González-Solís *et al.* (2009)
Cape Verde Shearwater	*Calonectris edwardsii*	CVS	34	Northern	NT	Cape Verde Islands	26	2006, 2007	González-Solís *et al.* (2009)

Smoothing cell size corresponds to the number of grid cells used for distribution smoothing in Zonation software for the analysis of habitat connectivity, and reflects the minimum area of habitat required for a species to occur. Cell sizes differ according to body size, following the approach of Leathwick *et al.* (2008). Each species was assigned to one of two assemblages based on its core distribution relative to the mean position of the northern boundary of the Subtropical Front (Burls & Reason, 2006). IUCN Status: LC, least concern; NT, near threatened; VU, vulnerable; EN, endangered; CR, critically endangered. Colony of the tracked animals is presented. Number of tracked individuals (N), tracked years and source of the tracking data.

For summer and winter periods, we removed bird locations from months when the number of points was below 10% of the maximum number for that species in any month in order to remove the interference of migratory movements from the seasonal data, but all available locations were included in the year-round models. Each species was classified as one of

two groups, based on its core distribution relative to the mean position of the northern boundary of the Subtropical Front (Burls & Reason, 2006): Southern species (i.e. species with a distribution largely south of 35°S within the area) and Northern species (i.e. species mainly distributed north of 35°S within the area). This categorization was based on previous evidence for segregation of seabird communities resulting mostly from latitudinal gradients in sea surface temperature (SST) (Péron *et al.*, 2010; Krüger & Petry, 2011; Navarro *et al.*, 2015; Quillfeldt *et al.*, 2015).

Environmental variables

Our species distribution models (SDMs) used 10-year average data (from summer, winter and year-round) for the following oceanographic variables: chlorophyll-*a* concentration (CHL), CHL anomaly, CHL gradients, SST, SST anomaly, SST gradients, sea surface height (SSH), SSH anomaly, wind speed, water depth and minimum distance to coast (DCOA) (Supporting Information Fig. S1). All variables, except DCOA, anomalies and gradients, were downloaded as monthly mean composite raster images from NOAA Coast-Watch Browser (http://coastwatch.pfeg.noaa.gov/). SST anomalies and CHL anomalies for each month were calculated as the difference between the average value for a given month and year, and the average for that month over a 10-year period in that grid cell. Gradients were generated by calculating the standard deviation of each cell in relation to the adjacent cell values (Sidhu *et al.*, 2012; Li *et al.*, 2015). All raster files were processed in ArcMap 10.2.

Species distribution models

Tracking data were combined with environmental predictors to generate SDMs from presence-only data using the Maximum Entropy (MaxEnt) software (Phillips, Anderson & Schapire, 2006b; Oppel *et al.*, 2012; Quillfeldt *et al.*, 2013) on a 1° × 1° spatial grid (e.g. Pinet *et al.*, 2011; Ramírez *et al.*, 2013; Missagia *et al.*, 2015), as the mean error of geolocation tags is usually lower than 1° (Wilson *et al.*, 1992; Phillips *et al.*, 2004; Nielsen & Sibert, 2007). Geolocators are useful to detect and model core areas for animals' distribution (Quillfeldt *et al.*, 2013; Krüger *et al.*, 2016, 2017) and compared to more accurate methods like GPS or Argos, geolocators tend to slightly inflate home range of animals (Phillips *et al.*, 2004). To compensate this potential inflation, we were conservative in the next steps of the analysis by only assuming areas of high importance values (see next section) to propose the key areas. Furthermore, geolocators are able to collect data for a longer period of time that is highly limited by battery size in GPS and PTTs, and this may also represent a barrier to sample distribution data on smaller seabirds like gadfly petrels and prions which cannot carry much weight. The tracking data were divided into training and test data by randomly setting aside 10% of the tracking dataset for spatial evaluation of the models (Araújo & Guisan, 2006; Austin, 2007). The first step of the modelling consisted of principal component analyses (PCA) to

eliminate any potential effect of multicollinearity. This is a recommended procedure as our goal was to model species distributions, rather than to verify relationships between species occurrence and the environment (Merow, Smith & Silander, 2013). We extracted the scores for components with eigenvalues above 1. With those scores, we ran MaxEnt on the presence-only positions, 50 times, with a bootstrap procedure to obtain a prediction of the average distribution (Edrén *et al.*, 2010). We assessed the accuracy of models using the area (AUC) under the receiver operating characteristic (ROC) curve. The AUC estimates the likelihood that a randomly selected presence point is located in a raster cell with a higher probability value for species occurrence than a randomly generated point (Phillips, *et al.*, 2006b).

Calculating area importance values

The outputs from MaxEnt were used to calculate area importance values for each bird assemblage using the Zonation software (Moilanen *et al.*, 2005; Moilanen & Wintle, 2006; Leathwick *et al.*, 2008). Zonation is assumed as one of the best programmes to set out conservation priorities when efforts focus on ecological communities and habitat connectivity (i.e. Blumentrath, 2011; Delavenne *et al.*, 2012). Zonation generates a raster with pixels representing the importance value, taking into account the probability of occurrence of each species given its weighting (see below) and response to habitat. The final output is scaled from 0 to 1, representing the least (0) to most important (1) areas. Each species was weighted by its IUCN conservation status as follows: least concern (LC) = 1; near threatened (NT) = 2; vulnerable (VU) = 3; endangered (EN) = 4 and critically endangered (CR) = 5. Boundary quality penalty curves (BQPCs; Moilanen & Wintle, 2007), which are measures of species responses to habitat connection, were constructed empirically based on the IUCN conservation status. We generated responses with changing slopes for CR = −0.01, EN = −0.008, VU = −0.004, NT = −0.002 and LC = 0. Thus, the proportion of area occupied by each species decreases with increasing habitat disconnection (except for LC) and the intensity of the decrease (slope) is proportional to the level of threat.

We evaluated the sensitivity of the weighting and habitat connectivity response (more details in Supporting Information, SI). Changing the values for these two variables had no substantial effect on the final output, except for a complete null model considering all species as equivalent, the output from which was unrealistic and did not match the species distributions (Supporting Information SI and Fig. S2–S4). Probability of occurrence in a given pixel diminishes when habitat characteristics differ from those within surrounding cells, using a home range-based grid size (Leathwick *et al.*, 2008). Uncertainty analysis (Moilanen & Wintle, 2006) was disabled in the computation (Leathwick *et al.*, 2008). The importance value was calculated for each assemblage during the three specified periods (summer, winter and year-round). We were conservative to propose key areas due geolocators errors (see previous section) and selected areas that fell

within 10% (90% of importance value) of the distribution threshold (Moilanen *et al.*, 2005), areas of 5% (95%) and 1% (99%) threshold (Oppel *et al.*, 2012), in a nested design – these areas are within the 10% threshold. The locations of the 1% candidate MPAs were compared with those of existing MPAs (Marine Conservation Institute, 2015), EBSAs (Dunn *et al.*, 2014) and mIBAs (Birdlife International, 2015).

Environmental threats

We examined spatial risk from five environmental threats: ocean acidification, oil pollution, floating plastics, shipping and fishing intensity (Supporting Information SII and, Fig. S5), which are environmental stressors largely recognized as stressors to which marine animals, particularly seabirds, are exposed and vulnerable, by means of spatial overlap and evidence of impact: acidification (Grémillet & Boulinier, 2009), oil pollution (Camphuysen & Heubeck, 2001; Wiese & Robertson, 2004; Moreno *et al.*, 2013), plastic (Titmus & Hyrenbach, 2011; Jiménez *et al.*, 2015; Wilcox *et al.*, 2015), shipping (Hatch *et al.*, 2008), fisheries (Anderson *et al.*, 2011; Croxall *et al.*, 2012). These variables were standardized to have a mean of 0 and an SD of 1 (Zuur, Ieno & Smith, 2007).

Cost analysis

We evaluated costs for the proposed key areas, using the layers from Fishing Vessels Density and overall Ship Density (Supporting Information SII and Fig. S5) as cost layers (Dalton, 2004; Leathwick *et al.*, 2008) in the Zonation software. We are using those layers as 'costs' because both are the most manageable factors from the most widespread human threats affecting marine fauna worldwide. Impact of fisheries on marine fauna are well described in literature and it includes food depletion through overfishing and direct mortality through bycatch (Becker & Beissinger, 2006; Cury *et al.*, 2011) but the vessel traffic is a potential source of oil (Halpern *et al.*, 2008, 2015; Hatch *et al.*, 2008) and noise (Morton & Symonds, 2002; Weilgart, 2007; Codarin *et al.*, 2009) pollution, and birds may collide against vessels during the night (Black, 2005; Glass & Ryan, 2013).

Zonation uses cost layers to reduce the value (importance to conservation based on the previously specified parameters) of a given cell, thus cost is interpreted as a variable that reduce the conservation value of a given area. The 'cost' variables were used here as a cost/limitation imposed to human activities, in the perspective that a no-cost represents no changes in the current activities, and a cost represent restrictions to activities. As our valuing of areas considers no influence of the human activities, sole the seabirds distribution regarding environmental variables, this represents the hypothetical value of the area under no constraint by human activities. Then it is possible to measure differences on the conservation value of an area when comparing different cost scenarios with the previous area valued under no influence of human activities. We used two cost constraint scenarios

to evaluate how the intensity of human activities may change the conservation value of candidate MPAs (i.e. key marine areas) if shipping and fishing were not regulated (no-cost, scenario 1) or partially regulated (cost, scenario 2). Cost constraints were calculated to each time frame (Summer, Winter and Year-round). The resulting values for each cost model were then subtracted from the respective area importance value within the 99, 95 and 90 threshold values (the proposed areas based on seabird distribution only), to evaluate how much each of the human activities change the conservation values within the proposed key areas.

Statistical analysis

We used the pixel value of the rasters as point information, and the centroid of longitude and latitude (N = 1697) as the geographical position of each grid cell. To evaluate the degree of correspondence between the distribution probabilities for the two seabird communities (MaxEnt outputs) and the distributions of the environmental threats, we used canonical correspondence analysis (CCA) using 'cca' function from the 'vegan' package (Oksanen *et al.*, 2013) within the R environment (R Core Team, 2015). Species probabilities were arcsine transformed. Species matrix entered the analysis as dependent matrix, and the threats entered the analysis as the independent matrix, with geographical coordinates used as covariables to control for spatial auto-correlation (cca[Species Matrix ~ Threat Matrix + Latitude × Longitude]). Each time frame was analysed separately. After running the analysis, we quantified the Euclidian distance of the species from the threats in the CCA bi-dimensional space to compare shifts in group exposition to threats.

We compared the change in value within the proposed key areas between cost models, time frames and assemblages through a Generalized Linear Mixed Model in 'lme4' package within R environment (Bates *et al.*, 2015). As threshold values for proposed key areas (90, 95 and 99% of importance values) are in a nested design, we entered the threshold as a random factor in the model.

Results

Species distribution models

Many of the environmental variables were highly correlated (Supporting Information Table S1). The power of the models using PC1 and PC2 to predict species distribution was high (AUC = 0.79 ± 0.06), although the models were less accurate for species with large latitudinal ranges, such as Cory's Shearwater *Calonectris borealis*, Cape Verde Shearwater *Calonectris edwardsii*, Great Shearwater *Ardenna gravis* and Wandering Albatross *Diomedea exulans*. It was clear that dividing the data into two seasons resulted in a slightly more accurate prediction of distributions than using data from the entire year (AUC summer = 0.80 ± 0.06; AUC winter = 0.83 ± 0.05; AUC year-round = 0.76 ± 0.06), probably because species occupy only part of the annual

distribution in any one season (Supporting Information Fig. S6–S8).

Assemblages and environmental change factors

Canonical correspondence analysis yielded slightly different results for each time frame, for Summer, the constrained analysis captured 74.1% of data variability (axis1 = 83.0%, axis2 = 12.3%), 70.7% for Winter (axis1 = 81.9%, axis2 = 15.9%) and 77.2% for all Year (axis1 = 81.9%, axis2 = 14.9%). Probability of occurrence of species (species matrices) was significantly matched by the distribution of threats for Summer ($F_{8,1638}$ = 585.49, P = 0.001), Winter ($F_{8,1638}$ = 494.48, P = 0.001) and all Year ($F_{8,1638}$ = 691.34, P = 0.001).

There was a clear segregation between assemblages within the bi-dimensional space of the CCA which showed that Northern species were more exposed to plastic pollution and acidification, while Southern species were exposed to Fisheries and Ship Traffic (Fig. 1a–c). However, such trend was not obvious in Winter (Fig. 1b) as the two Northern species (TA and TP) were highly pelagic and occurred in areas of

low cumulative threat occurrence (see Supporting Information Fig. S5 and S7). In Winter, two sub-groups of Southern seabirds were differentially exposed to acidification and plastic pollution (AP, GHA, WA) or to Fisheries and Ship Traffic (BBA, NGP, SGP). Two species were displaced from threat influence in two cases (MS and WCP) which seem to be caused by their wide latitudinal distribution. This was reflected in the bi-dimensional distance of each group from threats, where Northern seabirds increased their distance from threats in Winter compared to Summer and all Year, but Southern seabirds decreased their distance from threats in Winter compared to Summer and all Year, with exception of WCP (Fig. 1d–i).

Proposed key marine areas

The areas of highest value for Southern species did not vary seasonally, and were off the southern tip of South America and near the Falkland Islands (Fig. 2). In contrast, results for Northern species were highly seasonal: (1) during summer, the highest value areas were off the central coast of Brazil, whereas in the winter, these extended to oceanic waters in the mid-south Atlantic, (2) considering the whole year, the
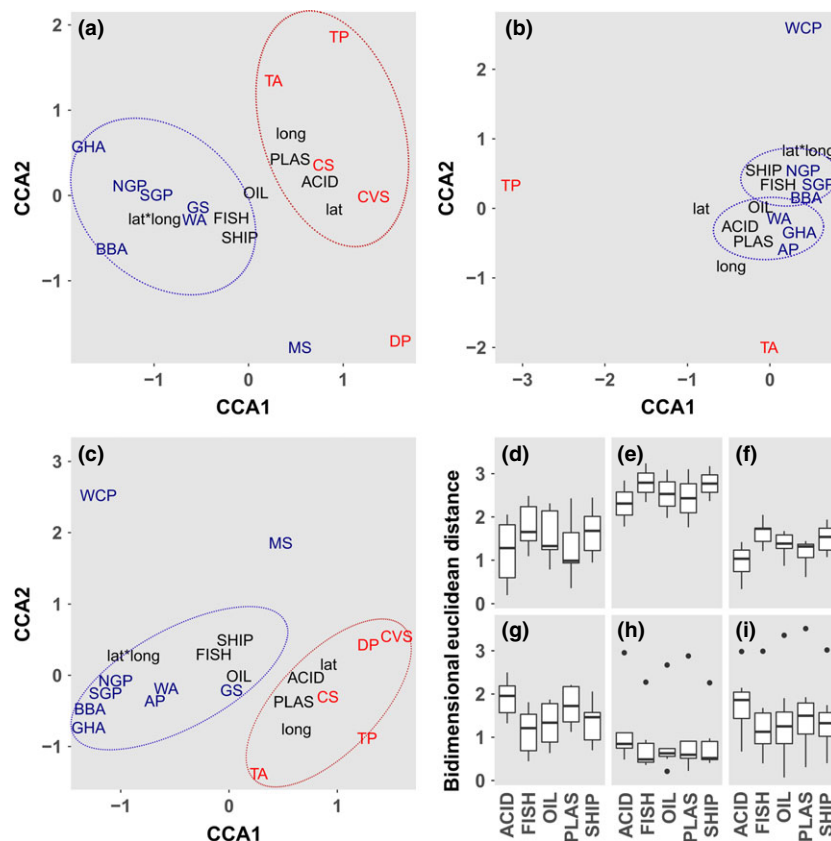

Figure 1 Canonical correspondence analysis bi-plots presenting the position of Northern species (red labels), Southern species (blue labels) and environmental – threats and coordinates – (black labels) for Summer (a), Winter (b) and all Year (c). Boxplots presenting the distribution frequency of the bi-dimensional distance of species from threats for Northern species during Summer (d), Winter (e) and all Year (f), and Southern species during Summer (g), Winter (h) and all Year (i). Top outliers in 'h' and 'i' are White-chinned Petrel values. [Colour figure can be viewed at wileyonlinelibrary.com]
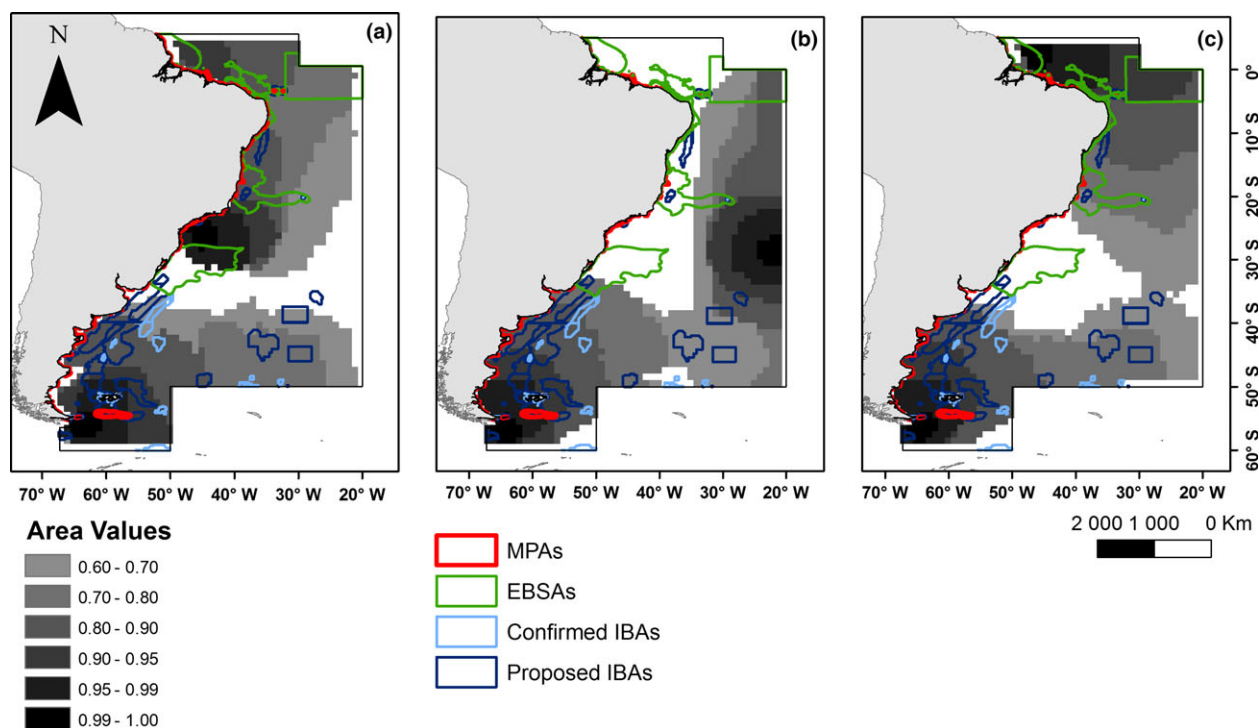
Figure 2 Zonation outputs in proportion to value based on the probability of occurrence of species given their weight and response to fragmentation for Summer (a), Winter (b) and year-round (c). Also shown the Official marine protected areas (MPAs, red lines), ecologically or biologically significant areas (EBSAs; green lines; https://www.cbd.int/ebsa/) and confirmed (light blue lines) or proposed (dark blue lines) BirdLife marine IBAs (http://maps.birdlife.org/marineIBAs/default.html). [Colour figure can be viewed at wileyonlinelibrary.com]

highest value areas were coastal and pelagic waters off northern Brazil (Fig. 2). The zonation value increased with the species occurrence probability (Fig. 3a, c, e) and with increased number of species (Fig. 3b, d, f). It means the high valued areas are a good representation for the occurrence of species.

The candidate protected areas based on the Southern assemblage were concentrated in the same area during summer, winter and year-round (Fig. 4). The three temperate areas were located south of Isla de los Estados and northwest of the Yaghan Basin. The candidate MPAs for Southern seabirds overlapped in the north with the current Isla de
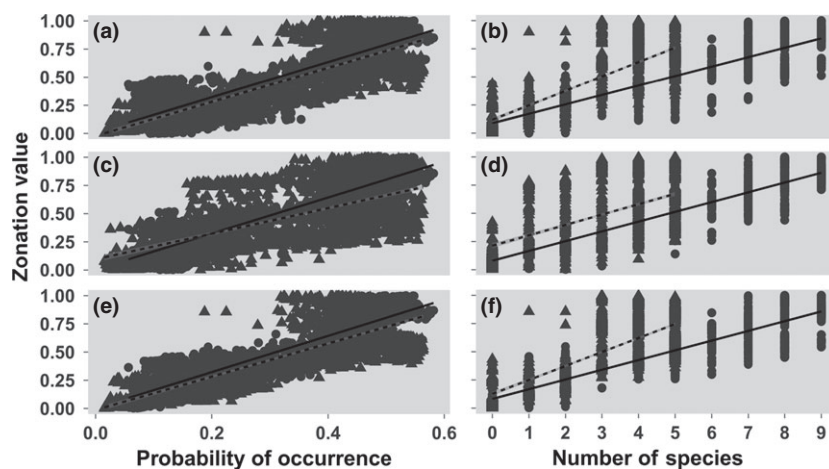

Figure 3 The zonation calculated area value in relation to the species occurrence probability (a, c, e) and number of species occurrence (b, d, f). Linear trend for Southern (solid line) and Northern (dashed line) seabird assemblages.
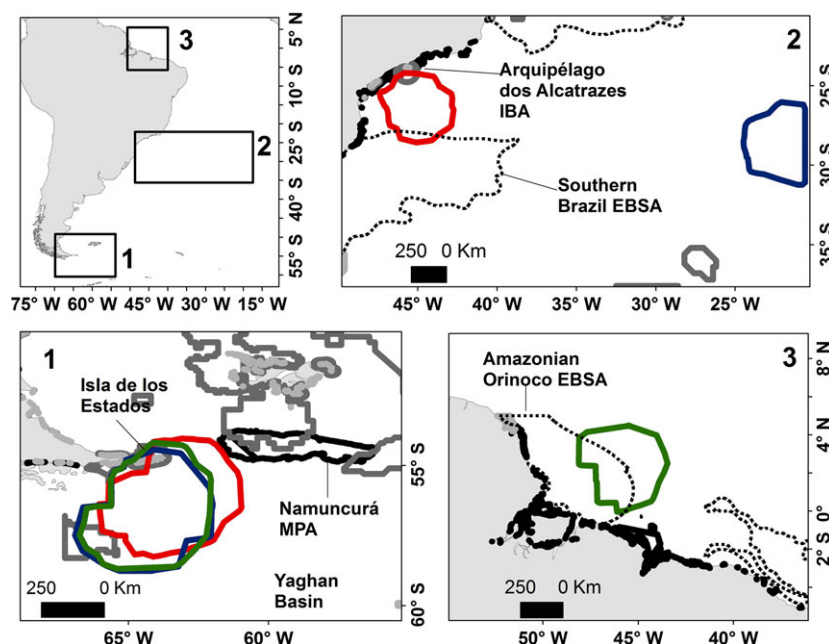
Figure 4 Key marine areas for 99% value threshold overlapped with Confirmed (light grey) or Proposed (dark grey) IBAs (http://maps.birdlife.org), ecologically or biologically significant marine areas (EBSAs; dashed line; https://www.cbd.int/ebsa/) and designated MPAs (black line; http://www.mpatlas.org). Southern (1), Northern (2 and 3) seabirds during summer (red line), winter (blue line) and all year (green line). [Colour figure can be viewed at wileyonlinelibrary.com]

los Estados MPA (Argentinean Ecological and Provincial Reserve) and IBA (proposed to protect BBA and Sooty Shearwater *Ardenna grisea*) and in the south with Southwest 33 IBA (proposed for GHA), but most areas of high value for Southern seabirds lacked any protection under current regimes (Fig. 4). On the other hand, the candidate protected areas for Northern seabirds were completely different among time frames. The candidate summer area on North overlapped with part of the proposed EBSA off southern Brazil (proposed due to occurrence of a strong upwelling) and the IBA on Arquipélago dos Alcatrazes (proposed based on the occurrence of Magnificent Frigatebirds *Fregata magnificens* and the Brown Boobies *Sula leucogaster*). Only a small fraction of a designated MPA overlapped the key area identified for Northern seabirds in the summer: the Litoral Centro Environmental Protection Area. During winter, the key area for Northern seabirds was in pelagic waters, over the gyre northeast of the Rio Grande Rise, where there is a large elevation of the seabed. Currently, no EBSA or IBA has been proposed in this region. Finally, the year-round key area for Northern seabirds was on the shelf slope and pelagic waters off northern Brazil, reaching as far as an oceanic ridge, where there is a strong upwelling from the Amazon River (Fig. 4). Roughly, one-third of this area overlapped with the Amazonian-Orinoco EBSA, proposed due to the enhanced marine productivity occurring within this area (Fig. 4). The overlap of our proposed key areas with existing MPAs was minimal (less than 1% Table 2). The percentage of mIBAs and EBSAs overlapped by our key areas was also minimal; however, overlap of Northern areas with EBSAs was

Table 2 Percentage of proposed key areas overlapping and overlapped by the IBAs, EBSAs and current MPAs

Assemblage	IBA confirmed	IBA proposed	EBSA	MPAs
Overlap with				
South	0.058	1.195	0.000	0.000
North	0.000	0.382	3.559	0.002
Overlapped by				
South	0.053	8.397	0.000	0.026
North	0.000	1.139	15.345	0.037

IBA, Important Bird Area; EBSA, ecologically and biologically significant area; MPA, marine protected area.

substantially greater than Southern areas, while Southern Areas were more overlapped by proposed mIBAs than Northern areas (Table 2).

Cost analysis

The use of cost constraints for Southern seabirds resulted in few spatial differences in values when using Fishing Density as a cost layer, but higher conservation values in pelagic areas shifted north when using Ship Density as a cost layer (Supporting Information Fig. S9). On the other hand, using Fishing Density as a cost layer for Northern seabirds displaced the zones of high values to spread south, and to spread to pelagic areas when Ship Density was the cost layer (Supporting Information Fig. S10).

The differences in both cost models were significant among time frames ($\chi^2_{26,6222} = 4143.9$, $P < 0.001$). The higher
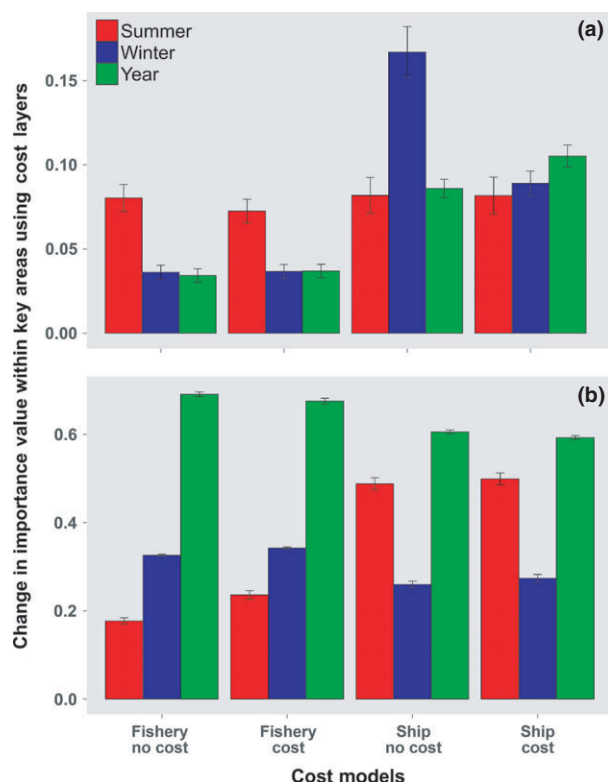
Figure 5 Change in values (proportion of reduction) within the proposed marine areas (thresholds 90, 95 and 99%) when using Fishing and Ship Densities as cost layers to calculate important values in Zonation. Southern (a) and Northern (b) seabirds. [Colour figure can be viewed at wileyonlinelibrary.com]

changes in values for Southern occurred when we used Fishing Density as a cost layer during summer, for both No Cost and Cost models, for winter, when using Ship Density No Cost model and for year, when using Ship Density Cost model (Fig. 5, Supporting Information Table S2). On the other hand, the change in values for Northern were higher for year-round for both No Cost and Cost models and for both Fishing and Ship Density cost layer, despite using Ship Density there was also a high change in value during summer for both cost models (Fig. 5, Supporting Information Table S2).

Discussion

In our study, we considered a modelling approach that distinguishes between ecological groups and accounts for seasonality in distribution of pelagic seabirds, with a view to identify a more representative network of key sites that would be more effective year-round MPAS. This approach located areas where currently there are few designated or proposed MPAs, suggesting that a community integrated approach can be an effective way to propose new pelagic MPAs (Yorio, 2009; Ronconi *et al.*, 2012; Thaxter *et al.*, 2012) in addition to proposals which use one or few attributes of one area, such as the presence of seamounts, reefs or the occurrence of one focal species (Arcos *et al.*, 2012; Grecian *et al.*, 2012; Dunn *et al.*, 2014).

High valued areas, threats and benefits

We placed the most important area for Southern seabird species within the southwest Atlantic at the southern Argentina shelf slope, and around the Falkland Islands. This area has long been recognized as important for marine predators and conservation of biodiversity (Croxall & Wood, 2002). This is reflected in the large number of proposed BirdLife International mIBAs (BirdLife International, 2015), and several designated coastal MPAs, including the large reserves of Namuncurá and Isla de losEstados (Argentine National Parks; Marine Conservation Institute, 2015). These waters are used by several and abundant seabird species during the breeding and non-breeding seasons, including birds from Patagonia (Yorio *et al.*, 1999; Yorio, 2009), Falkland Islands (Grémillet *et al.*, 2000; White *et al.*, 2002), South Georgia (Croxall & Wood, 2002; Croxall *et al.*, 2005; Phillips *et al.*, 2006a; Navarro *et al.*, 2015), Tristan da Cunha and Gough (Ronconi, Ryan & Ropert-Coudert, 2010; Reid *et al.*, 2013) and New Zealand (Nicholls *et al.*, 2002). Marine mammals also rely on this area during part of the year, including Fin *Balaenoptera physalus*, Sei *B. borealis*, Minke *B. acutorostrata*, Sperm *Physeter macrocephalus*, Southern Bottlenose *Hyperoodon planifrons* and Long-finned Pilot *Globicephala melas* whales, Hourglass *Lagenorhynchus cruciger*, Peale's *L. australis* and Commerson's *Cephalorhynchus commersonii* dolphins, Fur Seals *Arctocephalus* spp. and Southern Elephant Seals *Mirounga leonina* (White *et al.*, 2002). The attractiveness of this area for marine top predators results from the extensive, year-round upwelling that extends from sub-Antarctic to temperate waters, which supports a rich food web, including diverse communities of abundant squid and fishes (Acha *et al.*, 2004; Miloslavich *et al.*, 2011). As a consequence of such rich communities, both fishing vessels and seabirds target those areas (Grémillet *et al.*, 2000, 2008; Wakefield, Phillips & Belchier, 2012). In fact we found a higher 'exposition' of seabird to fisheries, which is well-known for the study area (Xavier *et al.*, 2004; Bugoni *et al.*, 2008; Jiménez *et al.*, 2010; Krüger *et al.*, 2017), and elsewhere (Anderson *et al.*, 2011; Lewison *et al.*, 2014). On the other hand, the vessel traffic is a potential source of threat for marine fauna, caused by oil (Halpern *et al.*, 2008, 2015; Hatch *et al.*, 2008) and noise (Morton & Symonds, 2002; Weilgart, 2007; Codarin *et al.*, 2009) pollution.

The different time frame approaches resulted in completely different value outputs for Northern seabird species, which reflects the seasonality of species present in the area throughout the year. This is mostly a consequence of migratory species such as Deserta's Petrel, Trindade Petrel and Cory's Shearwater. It also reflects the different types of habitats targeted by these species. The highly valued area during summer supports a highly diverse coastal ecosystem, including corals reefs (Roberts *et al.*, 2002) and mangroves (Polidoro *et al.*, 2010), and a relatively narrow shelf with several seamounts (Miloslavich *et al.*, 2011), supporting several taxa, particularly sharks, cetaceans and euphausiids (Tittensor *et al.*, 2010), and is intensively used by the species in this study, as evidenced by other authors (Dias *et al.*, 2011;

Catry *et al.*, 2013; Dias, Granadeiro & Catry, 2013; Ramírez *et al.*, 2013; Missagia *et al.*, 2015; Krüger *et al.*, 2016). During winter, the most important areas matched with an area of low biodiversity (Tittensor *et al.*, 2010) and low productivity near the South Atlantic tropical gyre. Short-term decreases in chlorophyll concentration within gyres due to climate shifts are a potential issue of concern, with implications for management (Gregg, Casey & McClain, 2005; Polovina, Howell & Abecassis, 2008; Irwin & Oliver, 2009). This area also overlaps with the non-breeding distribution of the Critically Endangered Tristan Albatross (Reid *et al.*, 2013), and the at-sea distribution during the breeding season and immediate post-breeding period of the Vulnerable Trindade Petrel (Krüger *et al.*, 2016). The important areas year-round matched the upwelling where the Amazon and Orinoco river plumes reach seamounts and islands, such as Fernando de Noronha and São Pedro and São Paulo (Kitchingman *et al.*, 2008; Miloslavich *et al.*, 2011). It is important to emphasize that the high valued (99%) year-round areas reflected more the suitable habitat for the species than the bulk of species distribution (i.e. González-Solís *et al.*, 2009; Ramírez *et al.*, 2013; Ramírez *et al.*, 2015), despite the lower values (0.90) contained the bulk distribution presented in those other studies. However, the 99% valued area is recognized as biologically important for the intense upwelling, which supports a biodiverse ecosystem (Tittensor *et al.*, 2010; Miloslavich *et al.*, 2011; Selig *et al.*, 2014), so highlighting the value of this area for conservation, which is used less frequently by the seabird species in this study.

Many of our study species ingest large amounts of plastic debris (Petry & Fonseca, 2002; Jiménez *et al.*, 2015), and the sensitivity of the Northern species to Plastic pollution is an evidence of that. The drifting model used in this study (Van Sebille *et al.*, 2012) assumes that plastic particles are carried by currents and accumulate in zones of lower current speed, mainly gyres (Van Sebille, 2015). Titmus & Hyrenbach (2011) found that some seabird species target those same areas of plastic concentration, because they tend to be characterized by a high occurrence of squid and flying fish (Titmus & Hyrenbach, 2011; Wilcox *et al.*, 2015). Our results provide circumstantial supporting evidence, but highlight the need for more studies to evaluate the overlap of seabird distribution with floating plastics, and the associated risk. Similarly, direct evaluations of the effects and implications of spatial variation in rates of ocean acidification for seabirds are lacking. Acidification is linked to enhanced levels of CO_2 in the atmosphere released by human activities (Cao & Caldeira, 2008). Severe decreases in pH may affect lower trophic levels in food webs (Cao & Caldeira, 2008; Pörtner & Peck, 2010; Hale *et al.*, 2011), with consequent impacts on top predators (Grémillet & Boulinier, 2009).

The assemblage approach to identify candidate MPAs revealed important areas missed in previous efforts to define both EBSAs (Dunn *et al.*, 2014) and BirdLife International mIBAs (Lascelles *et al.*, 2012; Ronconi *et al.*, 2012), although in other respects there was some overlap. As the currently designated MPAs are all coastal (Marine Conservation Institute, 2015), our analyses of tracking data from pelagic seabirds showed almost no overlap with areas that are presently protected by national legislation.

Potential costs

We found an assemblage and time frame differential change in costs that suggests that the use of seasonal protected areas could mean less impact on human activities to achieve high conservation value as proposed by Hyrenbach *et al.* (2000). For instance, by changing the fishing density by half, the changes for values within the key Southern areas did not change substantially in relation to the no-cost model. On the other hand, for Northern areas, the changes in value were relatively higher when applying cost layers, and for all the cost models, the non-seasonal approach always resulted in a higher change in values. That is probably a result of the higher seasonality for the species occurrence in the northern areas, whereas the occurrence of species in the south off Patagonia and sub-Antarctic waters is more constant throughout the year. Leathwick *et al.* (2008) also showed that changes in conservation value of an area compared between models with and without use of cost layers can vary from 5 to 10%, but in our case, the mean differences could reach as high as 50% for Northern seabirds.

Conclusions

By assigning seabirds to different assemblages, we were able to detect additional candidate areas for protection not recognized in previous marine spatial planning initiatives for FAO Region 41 in the southwest Atlantic Ocean. Hence, our results help guiding conservation decisions at both national and international levels, in terms of potential new MPAs and implementation of those already designated or proposed as EBSAs or mIBAs. Our analysis support the network of areas proposed as mIBAs in sub-Antarctic waters; the ultimate goal of BirdLife International is that these areas receive statutory protection (Birdlife International, 2015). We also call attention for the need to delineate protected areas in pelagic tropical waters both within the Brazilian EEZ, and in international waters. Those unprotected regions are important seasonally or year-round, and are subject to a wide array of Human-related threats.

While in most cases, the establishment of MPAs is aimed at improving management of fishing activities (Pitchford *et al.*, 2007; Stevenson *et al.*, 2013), several other factors may threaten species and ecosystems. Human activities, such as fishing, ship traffic and oiling, may be controllable to a large extent if there is a means of enforcing compliance with effective management regimes, but other threats associated with dynamic features of natural systems, such as ocean acidification and plastic pollution, present considerable challenges in the design and implementation of MPA networks (Conroy *et al.*, 2011).

The application of our approach to other oceans should reveal new areas to be incorporated in conservation networks. The benefits elsewhere would be highly dependent on the degree of seasonality in predator distributions, but it

seems likely there are parallel situations where areas are only used for part of the year by migratory species. We envisage that this technique can be used on finer temporal (e.g. incubating vs. chick rearing periods) and spatial (e.g. by the use of GPS or PTT devices) scales.

Acknowledgements

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Supporting information

Additional Supporting Information may be found in the online version of this article at the publisher's web-site:

Figure S1 Environmental variables used for the Species Distribution Modelling.

Figure S2 Zonation output showing the how much the values attributed to each pixel change by the weighting method used in building the (a) null model, (b) weight model #1, (c) weight model #2 and (d) the full model.

Figure S3 Regression between the full model and the new weight models. Lines are linear trends ±SE.

Figure S4 Relation between the overall species probability of occurrence and zonation value for each of the four models. Lines are cubic trend ± SE.

Figure S5 Environmental change variables: (a) ocean surface acidification (ACID) measured in terms of normalized changes in aragonite saturation state between pre-industrial and modern times (see Halpern et al 2008); (b) Risk of oil pollution (OIL) measured as a distance probability given the known points of oil spills or oil platforms from Halpern et al (2008) as value 1; (c) 10-year mean percentage of plastic pollution (PLASTIC) based on the particle drift model (Van Sebille et al 2012 2015); (d) ship density (SHIP) as the average abundance of vessels per grid cell (LuxSpace 2010); (e) fishing intensity (FISH) is a measure of density of fishing vessels.

Figure S6 Summer probability of occurrence of Black-browed Albatross (BBA), Cory's Shearwater (CS), Cape Verde Shearwater (CVS), Deserta's Petrel (DP), Grey-headed Albatross (GHA), Great Shearwater (GS), Manx Shearwater (MS), Northern Giant Petrel (NGP), Southern Giant Petrel (SGP), Tristan Albatross (TA), Trindade Petrel (TP), Wandering Albatross (WA), as estimated by MaxEnt models. Probability varies from 0 (dark blue) to 1 (dark red).

Figure S7 Winter probability of occurrence of Antarctic Prion (AP), Black-browed Albatross (BBA), Grey-headed Albatross (GHA), Northern Giant Petrel (NGP), Southern Giant Petrel (SGP), Tristan Albatross (TA), Trindade Petrel (TP), Wandering Albatross (WA), White-chinned Petrel (WCP), as estimated by MaxEnt models.Probability varies from 0 (dark blue) to 1 (dark red).

Figure S8 Year-round probability of occurrence of Antarctic Prion (AP), Black-browed Albatross (BBA), Cory's Shearwater (CS), Cape Verde Shearwater (CVS), Deserta's Petrel (DP), Grey-headed Albatross (GHA), Great Shearwater (GS), Manx Shearwater (MS), Northern Giant Petrel (NGP), Southern Giant Petrel (SGP), Tristan Albatross (TA), Trindade Petrel (TP), Wandering Albatross (WA), White-chinned Petrel (WCP), as estimated by MaxEnt models. Probability varies from 0 (dark blue) to 1 (dark red).

Figure S9 Zonation value for the different cost models, for southern seabirds.

Figure S10 Zonation value for the different cost models models, for northern seabirds.

Table S1 Loadings from the PCA.

Table S2 Generalized Linear Mixed Model results comparing change in values for the proposed key marine areas between seabird assemblages (Southern and Northern) and time-frames (summer, winter, year) when using different cost models (no cost and cost for Fishing Density and Ship Density).