

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

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

18 **Keywords:** Ecologically and Biologically Significant Areas; Important Bird Areas; seabird
19 tracking; species distribution modelling; threats; Zonation

20 **Introduction**

21 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
22 only <4% of the global ocean area is considered to be experiencing very low human impacts
23 (Halpern *et al.*, 2015). Hence, there is a need for a major international effort to protect and
24 maintain the functioning of marine ecosystems, through the implementation of Marine Protected
25 Areas (MPAs) (Rockström *et al.*, 2009; Kachelriess *et al.*, 2014). Despite a global agreement to
26 designate MPAs covering 10% of the global ocean (Secretariat of the Convention on Biological
27 Diversity, 2006), only 2.98% is currently protected, and <1% of the global ocean comprise ‘no-
28 take’ zones, where no harvesting of marine resources is allowed (Marine Conservation Institute
29 2015). Furthermore, few established MPAs effectively and integrally protect their targeted
30 species, so their value is likely to be even lower than expected (Jameson, Tupper & Ridley,
31 2002; Mora & Sale, 2011; Edgar *et al.*, 2014).

32
33 To truly protect the diversity of marine systems, the design of MPAs must account for
34 its location (Jameson *et al.*, 2002; Sundblad, Bergström & Sandström, 2011), connectivity with
35 other areas (Sundblad *et al.*, 2011) number and proportion of protected oceanographic features
36 (e.g. upwellings, seamounts; Sundblad *et al.*, 2011; Dunn *et al.*, 2014), species occurring there
37 and their seasonality (Hyrenbach, Forney & Dayton, 2000; Mann & Lazier, 2006), climatic
38 variability (Jameson *et al.*, 2002), isolation from human impacts (Rife *et al.*, 2013; Edgar *et al.*,
39 2014), human activities (Charles & Wilson, 2009; Edgar *et al.*, 2014; Mazor *et al.*, 2014) and,
40 socio-economic use of the area (Charles & Wilson, 2009). Most MPAs are designed to improve
41 fisheries management, i.e. to manage commercial fish stocks in a more sustainable way (e.g.
42 (Pitchford, Codling & Psarra, 2007; Horta e Costa *et al.*, 2013ab; Stevenson, Tissot & Walsh,
43 2013). Few studies have accounted for other environmental factors in the design of MPAs. For
44 instance, ship movements (Dalton, 2004; Halpern *et al.*, 2008), exposure to plastics (Lebreton,
45 Greer & Borrero, 2012; van Sebille, England & Froyland, 2012; Wilcox, Van Sebille &
46 Hardesty, 2015), oil pollution (Wiese & Robertson, 2004; Moreno *et al.*, 2013; Kark *et al.*,

47 2015), and ocean acidification (Harvey, Gwynn-Jones & Moore, 2013) present substantial
48 impacts to marine biodiversity, and should be considered in MPA design (Kelleher, 1999;
49 Pomeroy *et al.*, 2005).

50 Wide-ranging, pelagic top predators have been widely proposed as useful tools for
51 pointing out important areas which can add conservation value to MPA networks (Hooker &
52 Gerber, 2004; Sergio *et al.*, 2008; Ronconi *et al.*, 2012). Top-predators integrate factors
53 affecting seasonal variation in abundance and distribution of their lower trophic-level prey over
54 broad areas (Montevecchi *et al.*, 2012; Tancell *et al.*, 2012; Michael, Jahneke & Hyrenbach,
55 2014). Seabirds, in this context, are relatively easy to sample at their breeding colonies and to
56 track them at sea by using electronic devices, providing information on foraging movements at
57 various spatio-temporal scales (Phillips *et al.*, 2006a; Delord *et al.*, 2014; Thiebot *et al.*, 2014).
58 Several important marine areas have been identified based on the distribution and occurrence of
59 seabird species and proposed to integrate the current network of MPAs. The marine Important
60 Bird Areas (mIBAs) represent one of the most recent wide-ranging international efforts to
61 identify relevant areas for marine conservation (Lascelles *et al.*, 2012, 2016).

62 Here, we applied spatial distribution modelling to tracking data from 14 pelagic seabird
63 species in the southwest Atlantic Ocean (Food and Agriculture Organization, FAO, Fishing
64 Area 41). The use of tracking data is deliberately used to identify important areas in offshore
65 waters since many important “coastal” areas have been identified for MPAs or MIBAs through
66 other approaches and investigations of coastal seabirds. We identified key areas based solely on
67 the climatic and oceanographic variables irrespective of the distribution of threats, *a posteriori*
68 we quantified sensitivity of species to threats and quantified the costs to human activities within
69 those areas that are important for the birds, by imposing restrictions of activities within the areas
70 and calculating changes in conservation value. Such approach is justified by evidences that the
71 distribution of threats matches oceanographic conditions targeted by marine animals (Karpouzi,
72 Watson & Pauly, 2007; Hatch *et al.*, 2008; Titmus & Hyrenbach, 2011; Krüger *et al.*, 2016a).
73 Our objectives were: (1) identify important areas for pelagic seabirds throughout the year; (2)

74 quantify the sensitivity of the identified important areas for seabird assemblages to
75 environmental pressures; (3) compare our spatially-explicit results to the existing networks of
76 proposed or established protected areas identified by international organizations, and (4) carry
77 out a cost analysis evaluation of the implementation of seasonal vs non-seasonal protected areas.
78 FAO Fishing Area 41 encompasses a wide range of latitudes from tropical to Antarctic waters,
79 including a large array of habitats and oceanographic conditions, the largest continental shelf in
80 the Southern Hemisphere and a highly productive convergence zone that support a biodiversity-
81 rich ecosystem, and major fisheries (Vasconcellos & Csirke, 2011). Despite several proposals to
82 identify key areas for conservation in pelagic ecosystems, either in the form of Ecologically and
83 Biologically Significant Areas (EBSAs) (Dunn *et al.*, 2014) or mIBAs (Lascelles *et al.*, 2012),
84 the designated MPAs inside FAO41 are mostly restricted to coastal regions (Marine
85 Conservation Institute 2015, <http://www.mpatlas.org/explore/>). Thus, such network of MPAs
86 does not protect marine pelagic species and systems. In this sense, our analysis prioritizes
87 habitats used by pelagic seabird communities weighted by their seasonality and extinction risk
88 (which may protect other pelagic species) as pelagic habitats are underrepresented in the MPAs
89 network within FAO41. The key areas identified with this study may be seen as supporting the
90 already proposed MPAs as add-ons, which indentifies pelagic and seasonal areas that were not
91 considered by previous methods.

92 **Materials and methods**

93 **Tracking data and bird assemblages**

94 We used geolocator (Global Location Sensor or GLS) tracking data from 14 seabird species,
95 occurring at the southwest Atlantic Ocean during part or all of the annual cycle (Table 1).
96 Deployment and other details are provided elsewhere (Grémillet *et al.*, 2000; Croxall *et al.*,
97 2005; Phillips *et al.*, 2006a; González-Solís, Croxall & Afanasyev, 2008; González-Solís *et al.*,
98 2009; Mackley *et al.*, 2010; Quillfeldt *et al.*, 2013; Ramírez *et al.*, 2013; Reid *et al.*, 2013;
99 Missagia *et al.*, 2015, Krüger *et al.*, 2016a,b). The data were modelled in three different periods:
100 summer (Oct.-Mar.), winter (Apr.-Sep.) and year-round (Jan.-Dec.). For summer and winter

101 periods, we removed bird locations from months when the number of points was below 10% of
102 the maximum number for that species in any month in order to remove the interference of
103 migratory movements from the seasonal data, but all available locations were included in the
104 year-round models. Each species was classified as one of two groups, based on its core
105 distribution relative to the mean position of the northern boundary of the Subtropical Front
106 (Burls & Reason, 2006): Southern species (i.e. species with a distribution largely south of 35°S
107 within the area) and Northern species (i.e. species mainly distributed north of 35°S within the
108 area). This categorisation was based on previous evidence for segregation of seabird
109 communities resulting mostly from latitudinal gradients in sea surface temperature (Péron *et al.*,
110 2010; Krüger & Petry, 2011; Navarro *et al.*, 2015; Quillfeldt *et al.*, 2015).

111 **Environmental variables**

112 Our Species Distribution Models (SDMs) used 10-year average data (from summer, winter
113 and year-round) for the following oceanographic variables: chlorophyll-*a* concentration (CHL),
114 CHL anomaly, CHL gradients, sea surface temperature (SST), SST anomaly, SST gradients,
115 Sea Surface Height (SSH), SSH anomaly, wind speed, water depth, and minimum distance to
116 coast (DCOA) (Supporting Information Fig. S1). All variables, except DCOA, anomalies and
117 gradients, were downloaded as monthly mean composite raster images from NOAA
118 CoastWatch Browser (<http://coastwatch.pfeg.noaa.gov/>). SST anomalies and CHL anomalies
119 for each month were calculated as the difference between the average value for a given month
120 and year, and the average for that month over a 10-year period in that grid cell. Gradients were
121 generated by calculating the standard deviation of each cell in relation to the adjacent cell values
122 (Sidhu *et al.*, 2012; Li *et al.*, 2015). All raster files were processed in ArcMap 10.2.

123 **Species Distribution Models**

124 Tracking data was combined with environmental predictors to generate SDMs from presence-
125 only data using the Maximum Entropy (MaxEnt) software (Opiel *et al.*, 2012; Quillfeldt *et al.*,
126 2013) on a 1°x1° spatial grid (e.g. Pinet *et al.*, 2011; Ramírez *et al.*, 2013; Missaglia *et al.*, 2015),
127 as the mean error of geolocation tags is usually lower than 1° (Wilson *et al.*, 1992; Phillips *et*

128 *al.*, 2004; Nielsen & Sibert, 2007). Geolocators are useful to detect and model core areas for
129 animals' distribution (Quillfeldt *et al.*, 2013; Krüger *et al.*, 2016a, 2016b) and compared to more
130 accurate methods like GPS or Argos, geolocators tend to slightly inflate home range of animals
131 (Phillips *et al.*, 2004). To compensate this potential inflation, we were conservative in the next
132 steps of the analysis by only assuming areas of high importance values (see next section) to
133 propose the key areas. Furthermore, geolocators are able to collect data for a longer period of
134 time that is highly limited by battery size in GPS and PTTs, and this may also represent a barrier
135 to sample distribution data on smaller seabirds like gadfly petrels and prions which can not
136 carry much weight. The tracking data were divided into training and test data by randomly
137 setting aside 10% of the tracking dataset for spatial evaluation of the models (Araújo & Guisan,
138 2006; Austin, 2007). The first step of the modelling consisted of principal component analyses
139 (PCA) to eliminate any potential effect of multicollinearity. This is a recommended procedure
140 as our goal was to model species distributions, rather than to verify relationships between
141 species occurrence and the environment (Merow, Smith & Silander, 2013). We extracted the
142 scores for components with eigenvalues above 1. With those scores, we ran MaxEnt on the
143 presence-only positions, 50 times, with a bootstrap procedure to obtain a prediction of the
144 average distribution (Edrén *et al.*, 2010). We assessed the accuracy of models using the area
145 (AUC) under the Receiver Operating Characteristic (ROC) curve. The AUC estimates the
146 likelihood that a randomly selected presence point is located in a raster cell with a higher
147 probability value for species occurrence than a randomly generated point (Phillips, Anderson &
148 Schapire, 2006).

149 **Calculating area importance values**

150 The outputs from MaxEnt were used to calculate area importance values for each bird
151 assemblage using the Zonation software (Moilanen *et al.*, 2005; Moilanen & Wintle, 2006;
152 Leathwick *et al.*, 2008). Zonation is assumed as one of the best programs to set out conservation
153 priorities when efforts focus on ecological communities and habitat connectivity (i.e.,
154 Blumentrath, 2011; Delavenne *et al.*, 2012). Zonation generates a raster with pixels representing

155 the importance value, taking into account the probability of occurrence of each species given its
156 weighting (see below) and response to habitat. The final output is scaled from 0-1, representing
157 the least (0) to most important (1) areas. Each species was weighted by its IUCN conservation
158 status as follows: Least Concern (LC) = 1; Near Threatened (NT) = 2; Vulnerable (VU) = 3;
159 Endangered (EN) = 4 and Critically endangered (CR) = 5. Boundary Quality Penalty Curves
160 (BQPCs; Moilanen & Wintle, 2007), which are measures of species responses to habitat
161 connection, were constructed empirically based on the IUCN conservation status. We generated
162 responses with changing slopes for CR = -0.01, EN = -0.008, VU = -0.004, NT = -0.002 and LC
163 = 0. Thus, the proportion of area occupied by each species decreases with increasing habitat
164 disconnection (except for LC) and the intensity of the decrease (slope) is proportional to the
165 level of threat.

166 We evaluated the sensitivity of the weighting and habitat connectivity response (more
167 details in Supporting Information SI). Changing the values for these two variables had no
168 substantial effect on the final output, except for a complete null model considering all species as
169 equivalent, the output from which was unrealistic and did not match the species distributions
170 (Supporting Information Fig. S2, S3, S4). Probability of occurrence in a given pixel diminishes
171 when habitat characteristics differ from those within surrounding cells, using a home range-
172 based grid size (Leathwick *et al.*, 2008). Uncertainty analysis (Moilanen & Wintle, 2006) was
173 disabled in the computation (Leathwick *et al.*, 2008). The importance value was calculated for
174 each assemblage during the three specified periods (summer, winter and year-round). We were
175 conservative to propose key areas due geolocators errors (see previous section) and selected
176 areas that fell within 1% (90% of importance value) of the distribution threshold (Moilanen *et*
177 *al.*, 2005), areas of 0.5% (95%) and 0.1 % (99%) threshold (Oppel *et al.*, 2012), in a nested
178 design – these areas are within the 1% threshold. The locations of the 0.1 % candidate MPAs
179 were compared with those of existing MPAs (Marine Conservation Institute 2015), EBSAs
180 (Dunn *et al.*, 2014) and mIBAs (Birdlife International 2015).

181 **Environmental threats**

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

191 **Cost analysis**

192 We evaluated costs for the proposed key areas, using the layers from Fishing Vessels Density
193 and overall Ship Density (Supporting Information SII and Fig. S5) as cost layers (Dalton, 2004;
194 Leathwick *et al.*, 2008) in the Zonation software. We are using those layers as ‘costs’ because
195 both are the most manageable factors from the most widespread human threats affecting marine
196 fauna worldwide. Impact of fisheries on marine fauna are well described in literature, like food
197 depletion through overfishing and direct mortality through bycatch (Becker & Beissinger, 2006;
198 Cury *et al.*, 2011) but the vessel traffic is a potential source of oil (Halpern *et al.*, 2008, 2015;
199 Hatch *et al.*, 2008) and noise (Morton & Symonds, 2002; Weilgart, 2007; Codarin *et al.*, 2009)
200 pollution, and birds may collide against vessels during the night (Black, 2005; Glass & Ryan,
201 2013).

202 Zonation uses cost layers to reduce the value (importance to conservation based on the
203 previously specified parameters) of a given cell, thus cost is interpreted as a variable that reduce
204 the conservation value of a given area. The “cost” variables were used here as a cost/limitation
205 imposed to human activities, in the perspective that a no-cost represent no changes in the current
206 activities, and a cost represent restrictions to activities. As our valuing of areas considers no
207 influence of the human activities, sole the seabirds distribution regarding environmental
208 variables, this represents the hypothetical value of the area under no constraint by human

209 activities. Then it is possible to measure differences on the conservation value of an area when
210 comparing different cost scenarios with the previous area valued under no influence of human
211 activities. We used two cost constraint scenarios to evaluate how the intensity of human
212 activities may change the conservation value of candidate MPAs (i.e. key marine areas) if
213 shipping and fishing were not regulated (no-cost, scenario 1) or partially regulated (cost,
214 scenario 2). Costs constraints were calculated to each time-frame (Summer, Winter and Year-
215 round). The resulting values for each cost model were then subtracted from the respective area
216 importance value within the 99, 95 and 90 threshold values (the proposed areas based on seabird
217 distribution only), to evaluate how much each of the human activities change the conservation
218 values within the proposed key areas.

219 **Statistical Analysis**

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

232 We compared the change of value within the proposed key areas between cost models,
233 time-frames and assemblages through a Generalized Linear Mixed Model in 'lme4' package
234 within R-environment (Bates *et al.*, 2015). As threshold values for proposed key areas (90%,

235 95% and 99% of importance values) are in a nested design, we entered the threshold as a
236 random factor in the model.

237 **Results**

238 **Species Distribution Models**

239 Many of the environmental variables were highly correlated (Supporting Information Table S1).
240 The power of the models using PC1 and PC2 to predict species distribution was high
241 (AUC=0.79 ± 0.06), although the models were less accurate for species with large latitudinal
242 ranges, such as Cory's Shearwater (*Calonectris borealis*), Cape Verde Shearwater (*Calonectris*
243 *edwardsii*), Great Shearwater (*Ardenna gravis*) and Wandering Albatross (*Diomedea exulans*).
244 It was clear that dividing the data into two seasons resulted in a slightly more accurate
245 prediction of distributions than using data from the entire year (AUC summer = 0.80 ± 0.06;
246 AUC winter = 0.83 ± 0.05; AUC year-round = 0.76 ± 0.06), probably because species occupy
247 only part of the annual distribution in any one season (Fig. S6, S7, S8).

248 **Assemblages and Environmental Change Factors**

249 Canonical Correspondence Analysis yielded slight different results for each time frame, for
250 Summer the constrained analysis captured 74.1% of data variability (axis1= 83.0%, axis2=
251 12.3%), 70.7% for Winter (axis1= 81.9%, axis2 = 15.9%) and 77.2% for all Year (axis1 =
252 81.9%, axis2 = 14.9%). Probability of occurrence of species (species matrices) was significantly
253 matched by the distribution of threats for Summer ($F_{8,1638}=585.49$, $P=0.001$), Winter ($F_{8,1638}=
254 494.48$, $P=0.001$) and all Year ($F_{8,1638}= 691.34$, $P=0.001$).

255 There was a clear segregation between assemblages within the bi-dimensional space of
256 the CCA which showed that Northern species were more exposed with plastic pollution and
257 acidification, while Southern species were associated with Fisheries and Ship Traffic (Fig.
258 1a,b,c). However, such trend was not obvious in Winter (Fig.1b) as the two Northern species
259 (TA and TP) were highly pelagic and occurred in areas of low cumulative threat occurrence (see
260 Supporting Information Fig. S5, Fig. S7). In Winter two sub-groups of Southern seabirds were

261 differentially exposed to Acidification and Plastic Pollution (AP, GHA, WA) or to Fisheries and
262 Ship Traffic (BBA, NGP, SGP). Two species were displaced from threat influence in two cases
263 (MS and WCP) which seem to be caused by their wide latitudinal distribution. This was
264 reflected in the bi-dimensional distance of each group from threats, where Northern seabirds
265 increased their distance from threats in Winter comparing to Summer and all Year, but Southern
266 seabirds decreased their distance from threats in Winter comparing to Summer and all Year,
267 with exception of WCP (Fig. 1d-i).

268 **Proposed key marine areas**

269 The areas of highest value for Southern species did not vary seasonally, and were off the
270 southern tip of South America and near the Falkland Islands (Fig.2). In contrast, results for
271 Northern species were highly seasonal: a) during summer, the highest value areas were off the
272 central coast of Brazil, whereas in the winter, these extended to oceanic waters in the mid-south
273 Atlantic, b) considering the whole year, the highest value areas were coastal and pelagic waters
274 off northern Brazil (Fig.2). The zonation value increased with the species occurrence probability
275 (Fig. 3a,c,e) and with increased number of species (Fig. 3b,d,f). It means the high valued areas
276 are a good representation for the occurrence of species.

277 The candidate protected areas based on the Southern assemblage were concentrated in
278 the same area during summer, winter and year-round (Fig.4). The three temperate areas were
279 located south of Isla de los Estados and northwest of the Yaghan Basin. The candidate MPAs
280 for Southern seabirds overlapped in the north with the current Isla de los Estados MPA
281 (Argentinean Ecological and Provincial Reserve) and IBA (proposed to protect BBA and Sooty
282 Shearwater *Ardenna grisea*) and in the south with Southwest 33 IBA (proposed for GHA), but
283 most areas of high value for Southern seabirds lacked any protection under current regimes
284 (Fig.4). On the other hand, the candidate protected areas for Northern seabirds were completely
285 different among time-frames. The candidate summer area on North overlapped with part of the
286 proposed EBSA off southern Brazil (proposed due to occurrence of a strong upwelling) and the
287 IBA on Arquipélago dos Alcatrazes (proposed based on the occurrence of Magnificent

288 Frigatebirds *Fregata magnificens* and the Brown Boobies *Sula leucogaster*). Only a small
289 fraction of a designated MPA overlapped the key area identified for Northern seabirds in the
290 summer: the Litoral Centro Environmental Protection Area. During winter, the key area for
291 Northern seabirds was in pelagic waters, over the gyre northeast of the Rio Grande Rise, where
292 there is a large elevation of the seabed. Currently no EBSA or IBA has been proposed in this
293 region. Finally, the year-round key area for Northern seabirds was on the shelf slope and pelagic
294 waters off northern Brazil, reaching as far as an oceanic ridge, where there is a strong upwelling
295 from the Amazon River (Fig.4). Roughly, one-third of this area overlapped with the
296 Amazonian-Orinoco EBSA, proposed due to the enhanced marine productivity occurring within
297 this area (Fig.4). The overlap of our proposed key areas with existing MPAs was minimal (less
298 than 1% Table 2). The percentage of mIBAs and EBSAs overlapped by our key areas was also
299 minimal, however overlap of Northern areas with EBSAs was substantially greater than
300 Southern areas, while Southern Areas were more overlapped by proposed mIBAs than Northern
301 areas (Table 2).

302 **Cost analysis**

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

309 The differences of both cost models were significant among Time-Frames ($\chi^2_{26,6222}=$
310 4143.9, $P<0.001$). The higher changes in values for Southern occurred when we used Fishing
311 Density as a cost layer during summer, for both No Cost and Cost models, for winter when
312 using Ship Density No Cost model and for year when using Ship Density Cost model (Fig.5,
313 Supporting Information Table S2). On the other hand, the change in values for Northern were
314 higher for year-round for both No Cost and Cost models and for both Fishing and Ship Density

315 cost layer, despite using Ship Density there was also a high change of value during summer for
316 both cost models (Fig.5, Supporting Information Table S2).

317 **Discussion**

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

326 **High Valued Areas, Threats and Benefits**

327 We placed the most important area for Southern seabird species within the southwest Atlantic at
328 the southern Argentina shelf slope, and around the Falkland Islands. This area has long been
329 recognized as important for marine predators and conservation of biodiversity (Croxall &
330 Wood, 2002). This is reflected in the large number of proposed BirdLife International mIBAs
331 (BirdLife International 2015), and several designated coastal MPAs, including the large reserves
332 of Namuncurá and Isla de los Estados (Argentine National Parks; Marine Conservation Institute
333 2015). These waters are used by several and abundant seabird species during the breeding and
334 non-breeding seasons, including birds from Patagonia (Yorio *et al.*, 1999; Yorio, 2009),
335 Falkland Islands (Grémillet *et al.*, 2000; White *et al.*, 2002), South Georgia (Croxall & Wood,
336 2002; Croxall *et al.*, 2005; Phillips *et al.*, 2006a; Navarro *et al.*, 2015), Tristan da Cunha and
337 Gough (Ronconi, Ryan & Ropert-Coudert, 2010; Reid *et al.*, 2013), and New Zealand (Nicholls
338 *et al.*, 2002). Marine mammals also rely on this area during part of the year, including Fin
339 (*Balaenoptera physalus*), Sei (*B. borealis*), Minke (*B. acutorostrata*), Sperm (*Physeter*
340 *macrocephalus*), Southern Bottlenose (*Hyperoodon planifrons*) and Long-finned Pilot
341 (*Globicephala melas*) whales, Hourglass (*Lagenorhynchus cruciger*), Peale's (*L. australis*) and

342 Commerson's (*Cephalorhynchus commersonii*) dolphins, Fur Seals (*Arctocephalus* spp.) and
343 Southern Elephant Seals (*Mirounga leonina*) (White *et al.*, 2002). The attractiveness of this area
344 for marine top predators results from the extensive, year-round upwelling that extends from sub-
345 Antarctic to temperate waters, which supports a rich food web, including diverse communities
346 of abundant squid and fishes (Acha *et al.*, 2004; Miloslavich *et al.*, 2011). As a consequence of
347 such rich communities, both fishing vessels and seabirds target those areas (Grémillet *et al.*,
348 2000, 2008; Wakefield, Phillips & Belchier, 2012). In fact we found a higher "exposition" of
349 seabird to fisheries, which is well-known for the study area (Xavier *et al.*, 2004; Bugoni *et al.*,
350 2008; Jiménez *et al.*, 2010, Krüger *et al.*, 2016a), and elsewhere (Anderson *et al.*, 2011;
351 Lewison *et al.*, 2014). In the other hand, the vessel traffic is a potential source of threat for
352 marine fauna, by oil (Halpern *et al.*, 2008, 2015; Hatch *et al.*, 2008) and noise (Morton &
353 Symonds, 2002; Weilgart, 2007; Codarin *et al.*, 2009) pollution.

354 The different time-frame approaches resulted in completely different value outputs for
355 Northern seabird species, which reflects the seasonality of species present in the area throughout
356 the year. This is mostly a consequence of migratory species such as Deserta's Petrel, Trindade
357 Petrel and Cory's Shearwater. It also reflects the different types of habitats targeted by these
358 species. The highly valued area during summer supports a highly diverse coastal ecosystem,
359 including corals reefs (Roberts *et al.*, 2002) and mangroves (Polidoro *et al.*, 2010), and a
360 relatively narrow shelf with several seamounts (Miloslavich *et al.*, 2011), supporting several
361 taxa, particularly sharks, cetaceans and euphausiids (Tittensor *et al.*, 2010), and is intensively
362 used by the species in this study, as evidenced by other authors (Dias *et al.*, 2011; Catry *et al.*,
363 2013; Dias, Granadeiro & Catry, 2013; Ramírez *et al.*, 2013; Missagia *et al.*, 2015; Krüger *et*
364 *al.*, 2016a). During winter, the most important areas matched with an area of low biodiversity
365 (Tittensor *et al.*, 2010) and low productivity near the South Atlantic tropical gyre. Short-term
366 decreases in chlorophyll concentration within gyres due to climate shifts are a potential issue of
367 concern, with implications for management (Gregg, Casey & McClain, 2005; Polovina, Howell
368 & Abecassis, 2008; Irwin & Oliver, 2009). This area also overlaps with the non-breeding

369 distribution of the Critically Endangered Tristan Albatross (Reid *et al.*, 2013), and the at-sea
370 distribution during the breeding season and immediate post-breeding period of the Vulnerable
371 Trindade Petrel (Krüger *et al.*, 2016b). The important areas year-round matched the upwelling
372 where the Amazon and Orinoco river plumes reach seamounts and islands, such as Fernando de
373 Noronha and São Pedro and São Paulo (Kitchingman *et al.*, 2008; Miloslavich *et al.*, 2011). It is
374 important to emphasize that the high valued (99%) year-round areas reflected more the suitable
375 habitat for the species than the bulk of species distribution (i.e. Gonzáles-Solís *et al.*, 2009,
376 Ramírez *et al.*, 2013, 2015), despite the lower values (0.90) embraced those areas presented by
377 those authors. However the 99% valued area is recognized as biologically important for the
378 intense upwelling, which supports a biodiverse ecosystem (Tittensor *et al.*, 2010; Miloslavich *et*
379 *al.*, 2011; Selig *et al.*, 2014), so highlighting the value of this area for conservation, which is used
380 less frequently by the seabird species in this study

381 Many of our study species ingest large amounts of plastic debris (Petry & Fonseca,
382 2002; Jiménez *et al.*, 2015), and the sensitivity of the Northern species to Plastic pollution
383 evidences that. The drifting model used in this study (Van Sebille *et al.*, 2012) assumes that
384 plastic particles are carried by currents and accumulate in zones of lower current speed, mainly
385 gyres (Van Sebille, 2015). Titmus & David Hyrenbach (2011) found that some seabird species
386 target those same areas of plastic concentration, because they tend to be characterised by a high
387 occurrence of squid and flying fish (Titmus & Hyrenbach, 2011; Wilcox *et al.*, 2015). Our
388 results provide circumstantial supporting evidence, but highlight the need for more studies to
389 evaluate the overlap of seabird distribution with floating plastics, and the associated risk.
390 Similarly, direct evaluations of the effects and implications of spatial variation in rates of ocean
391 acidification for seabirds are lacking. Acidification is linked to enhanced levels of CO₂ in the
392 atmosphere released by human activities (Cao & Caldeira, 2008). Severe decreases in pH may
393 affect lower trophic levels in food webs (Cao & Caldeira, 2008; Pörtner & Peck, 2010; Hale *et*
394 *al.*, 2011), with consequent impacts on top predators (Grémillet & Boulinier, 2009).

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

401 **Potential Costs**

402 We found an assemblage and time-frame differential change in costs that suggests that the use
403 of seasonal protected areas could mean less impact on human activities to achieve high
404 conservation value as proposed by Hyrenbach, Forney & Dayton (2000). For instance, by
405 changing the fishing density by half, the changes for values within the key Southern areas did
406 not change substantially in relation to the no-cost model. On the other hand, for Northern areas,
407 the changes in value were relatively higher when applying cost layers, and for all the cost
408 models the non-seasonal approach always resulted in a higher change in values. That's probably
409 a result of the higher seasonality for the species occurrence in the northern areas, whereas the
410 occurrence of species in the south off Patagonia and subantarctic waters is more constant
411 throughout the year. Leathwick *et al.* (2008) also showed that changes in conservation value of
412 an area compared between models with and without use of cost layers can vary from 5% to
413 10%, but in our case the mean differences could reach as high as 50% for Northern seabirds.

414 **Conclusions**

415 By assigning seabirds to different assemblages, we were able to detect additional candidate
416 areas for protection not recognised in previous marine spatial planning initiatives for FAO
417 Region 41 in the southwest Atlantic Ocean. Hence, our results help guiding conservation
418 decisions at both national and international levels, in terms of potential new MPAs and
419 implementation of those already designated or proposed as EBSAs or mIBAs. Our analysis
420 support the network of areas proposed as mIBAs in sub-Antarctic waters; the ultimate goal of
421 BirdLife International is that these areas receive statutory protection (BirdLife International

422 2015). We also call attention for the need to delineate protected areas in pelagic tropical waters
423 both within the Brazilian EEZ, and in international waters. Those unprotected regions are
424 important seasonally or year-round, and are subject to a wide array of Human-related threats.

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

432 The application of our approach to other oceans should reveal new areas to be
433 incorporated in conservation networks. The benefits elsewhere would be highly dependent on
434 the degree of seasonality in predator distributions, but it seems likely there are parallel situations
435 where areas are only used for part of the year by migratory species. We envisage that this
436 technique can be used on finer temporal (e.g. incubating versus chick rearing periods) and
437 spatial (e.g. by the use of GPS or PTT devices) scales.

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820

Table 1 Seabird species included in the study. 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 and Reason 2006): Temperate (TE) and tropical (TR) species. 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.

CommonName	Species	Abbreviation	Size Range (cm)	Assemblage	IUCN Status	Colony	N	Years	Source
Tristan Albatross	<i>Diomedea dabbenena</i>	TA	110	Northern	CR	Gough Island	34	2004, 2005, 2006	Reid et al (2013)
Wandering Albatross	<i>Diomedea exulans</i>	WA	120-135	Southern	VU	Bird Island (South Georgia)	18	2003	Mackley et al (2010)
Black-browed Albatross	<i>Thalassarche melanophris</i>	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	<i>Thalassarche chrysostoma</i>	GHA	70-85	Southern	EN	Bird Island (South Georgia)	35	2003, 2006	Croxallet al (2005)
Northern Giant Petrel	<i>Macronectes halli</i>	NGP	80-95	Southern	LC	Bird Island (South Georgia)	25	1999, 2001	González-Solis et al (2008)
Southern Giant Petrel	<i>Macronectes giganteus</i>	SGP	85-100	Southern	LC	Bird Island (South Georgia), Elephant Island (South Shetlands)	40	1999, 2001, 2011	González-Solis et al (2008), Krüger et al (2016)
White-chinned Petrel	<i>Procellaria aequinoctialis</i>	WCP	51-58	Southern	VU	South Georgia	11	2003, 2004	Phillips et al (2006)
Antarctic Prion	<i>Pachyptila desolata</i>	AP	25-27	Southern	LC	South Georgia	10	2009, 2010	Quillfeldt et al (2013)
Deserta's Petrel	<i>Pterodroma deserta</i>	DP	35	Northern	VU	Bugio Island	24	2007, 2008, 2009,	Ramírez et al (2013)

								2010		
Petrel	Trindade	<i>Pterodroma arminjoniana</i>	TP	37-40	Northern	VU	Trindade Island	4	2013,2014	Krüger et al (2016.)
Cory's Shearwater		<i>Calonectris borealis</i>	CS	46	Northern	LC	Berlenga, Azores and Canary Islands	41	2002, 2003, 2004, 2011, 2012	González-Solís et al (2009), Missaglia et al (2016)
Great Shearwater		<i>Ardenna gravis</i>	GS	46-51	Southern	LC	Gough Island	37	2008-2012	González-Solís
Manx Shearwater		<i>Puffinus puffinus</i>	MS	30-35	Southern	LC	Heimeaeey, Iceland	10	2006,2007	González-Solís et al (2009)
Cape Verde Shearwater		<i>Calonectris edwardsii</i>	CVS	34	Northern	NT	Cape Verde Isalnds	26	2006,2007	González-Solís et al (2009)

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

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

List of Figures

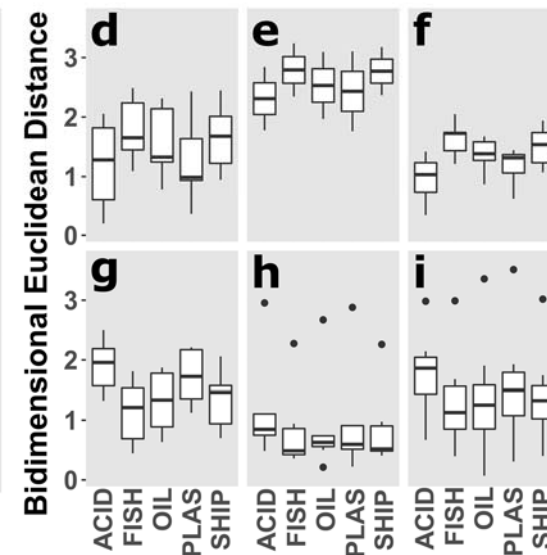
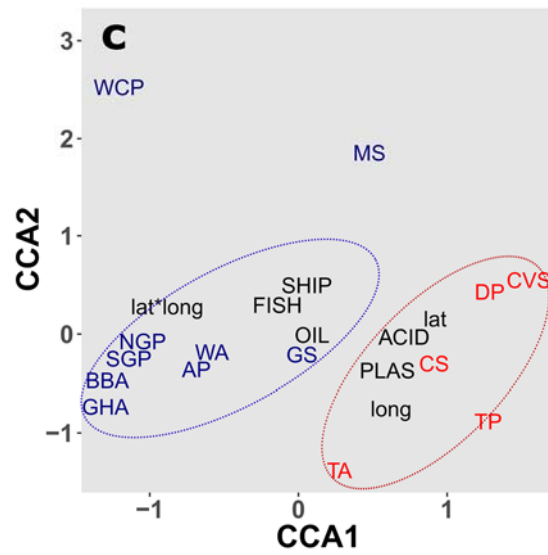
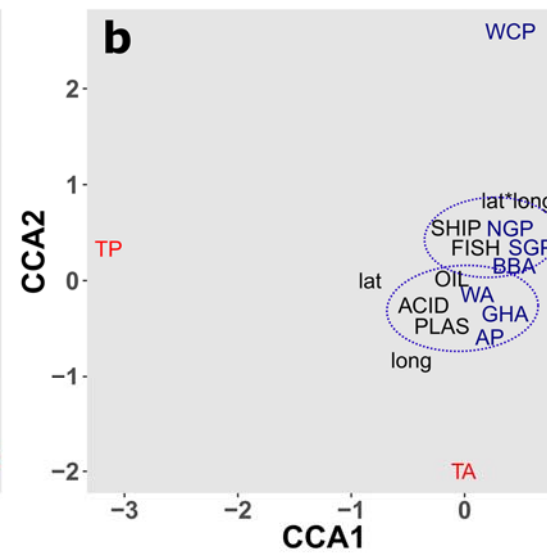
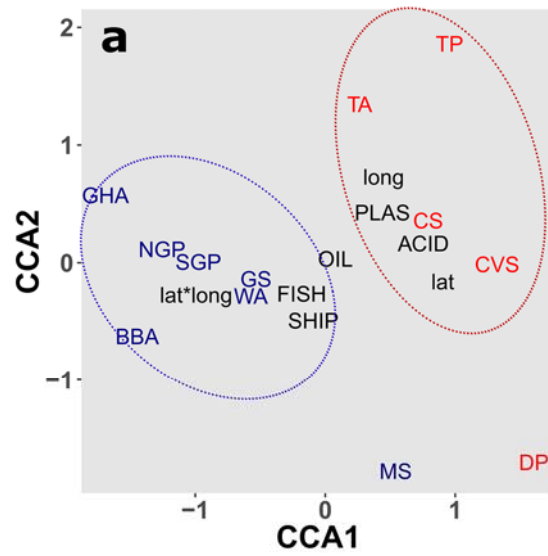
Figure 1 Canonical Correspondence Analysis bi-plots presenting the position of Northern species (red labels), Southern species (blue labels) and environmental – threats and coordinates – (black labels) for Summer (a), Winter (b) and all Year (c). Boxplots presenting the distribution frequency of the 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.

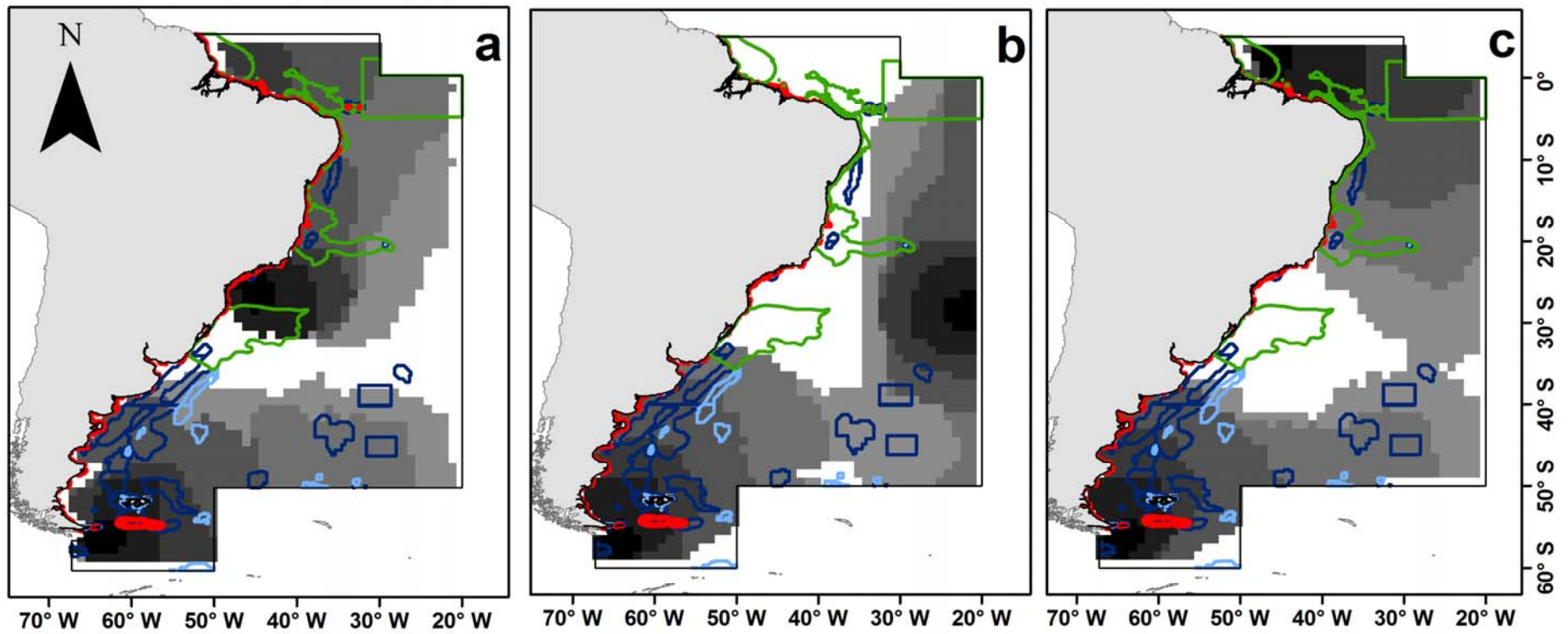
Figure 2 Zonation outputs in proportion to value based on the probability of occurrence of species given their weight and response to fragmentation for Summer (a), Winter (b) and year-round (c). Also shown the Official Marine Protected Areas (MPAs, red lines), Ecologically or Biologically Significant 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>).

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.

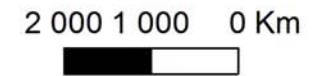
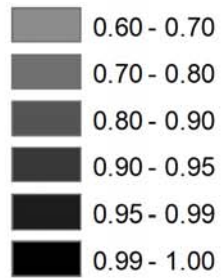
Figure 4 Key marine areas for 99% value threshold overlapped with Confirmed (light grey) or Proposed (dark grey) IBAs (<http://maps.birdlife.org>), Ecologically or Biologically Significant Marine Areas (EBSAs; dashed line; <https://www.cbd.int/ebsa/>) and designated MPAs (black line; <http://www.mpatlas.org>). Southern (1), Northern (2 and 3) seabirds during summer (red line), winter (blue line) and all year (green line).

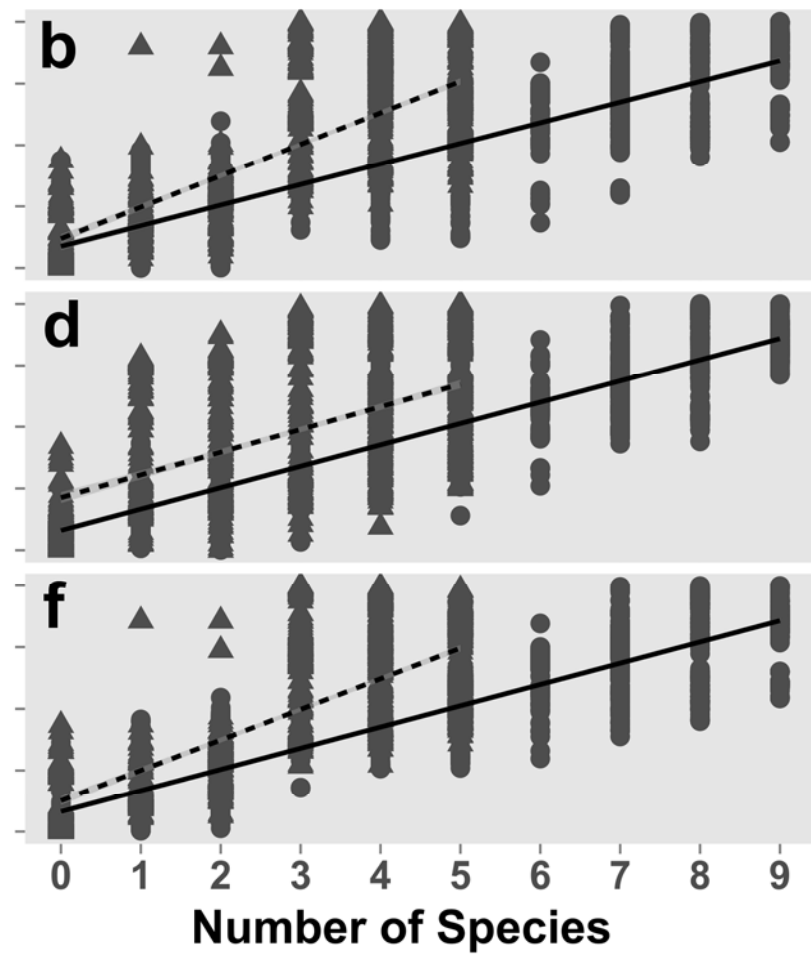
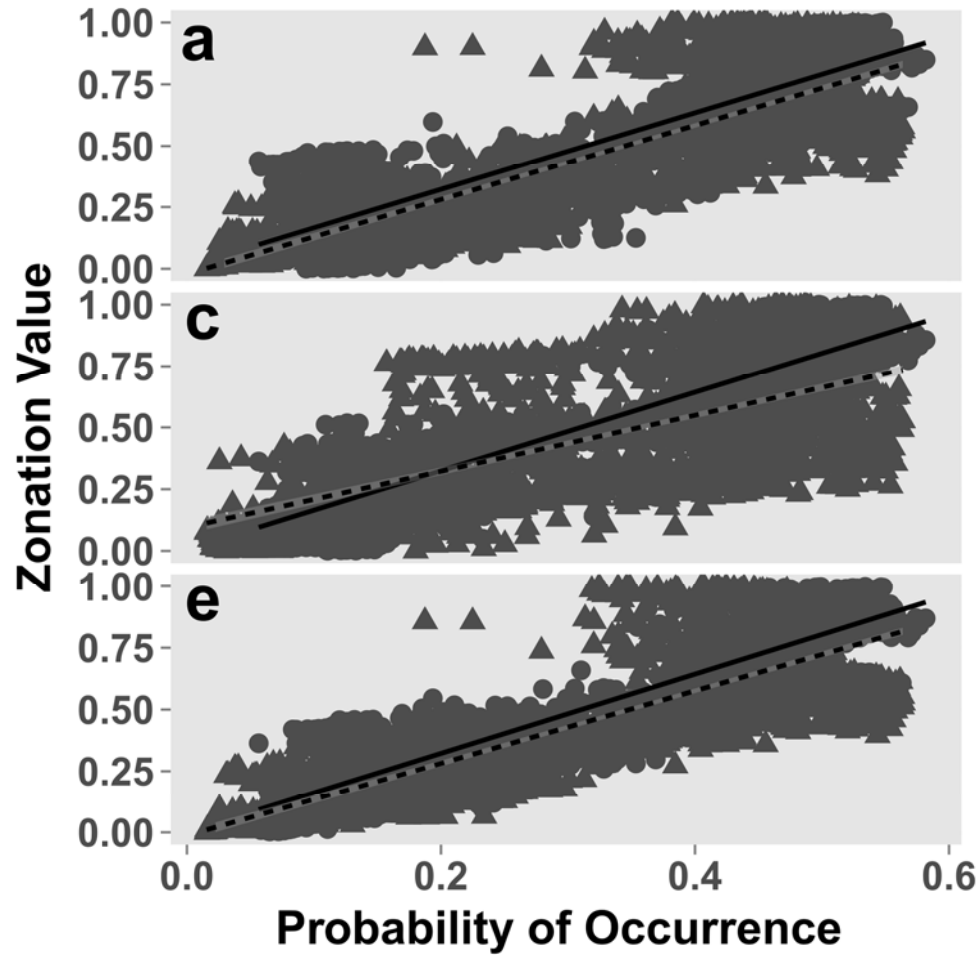
Figure 5 Change in values (proportion of reduction) within the proposed marine areas (thresholds 90%, 95% and 99%) when using Fishing and Ship Densities as cost layers to calculate importance values in Zonation. Southern (a) and Northern (b) seabirds.

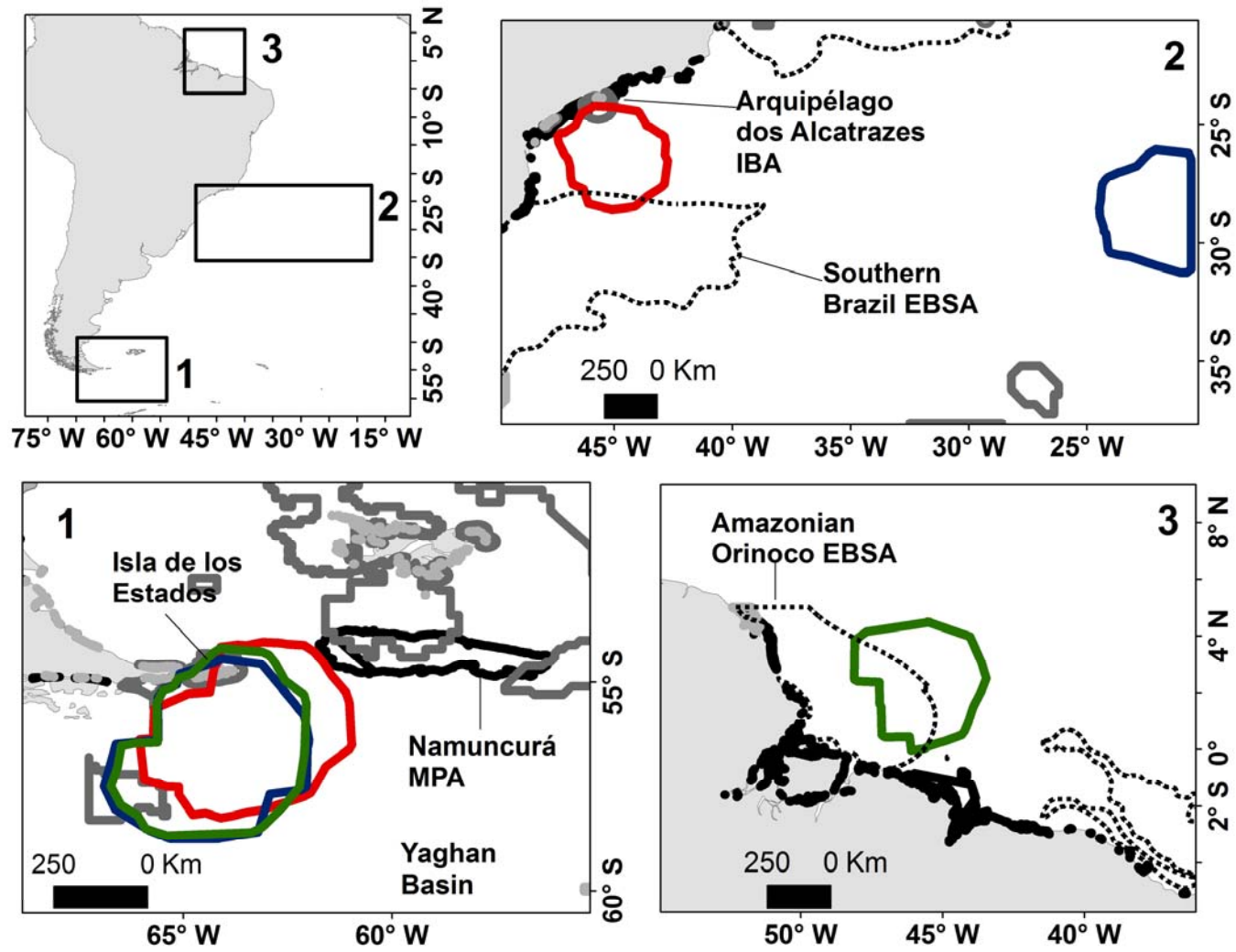


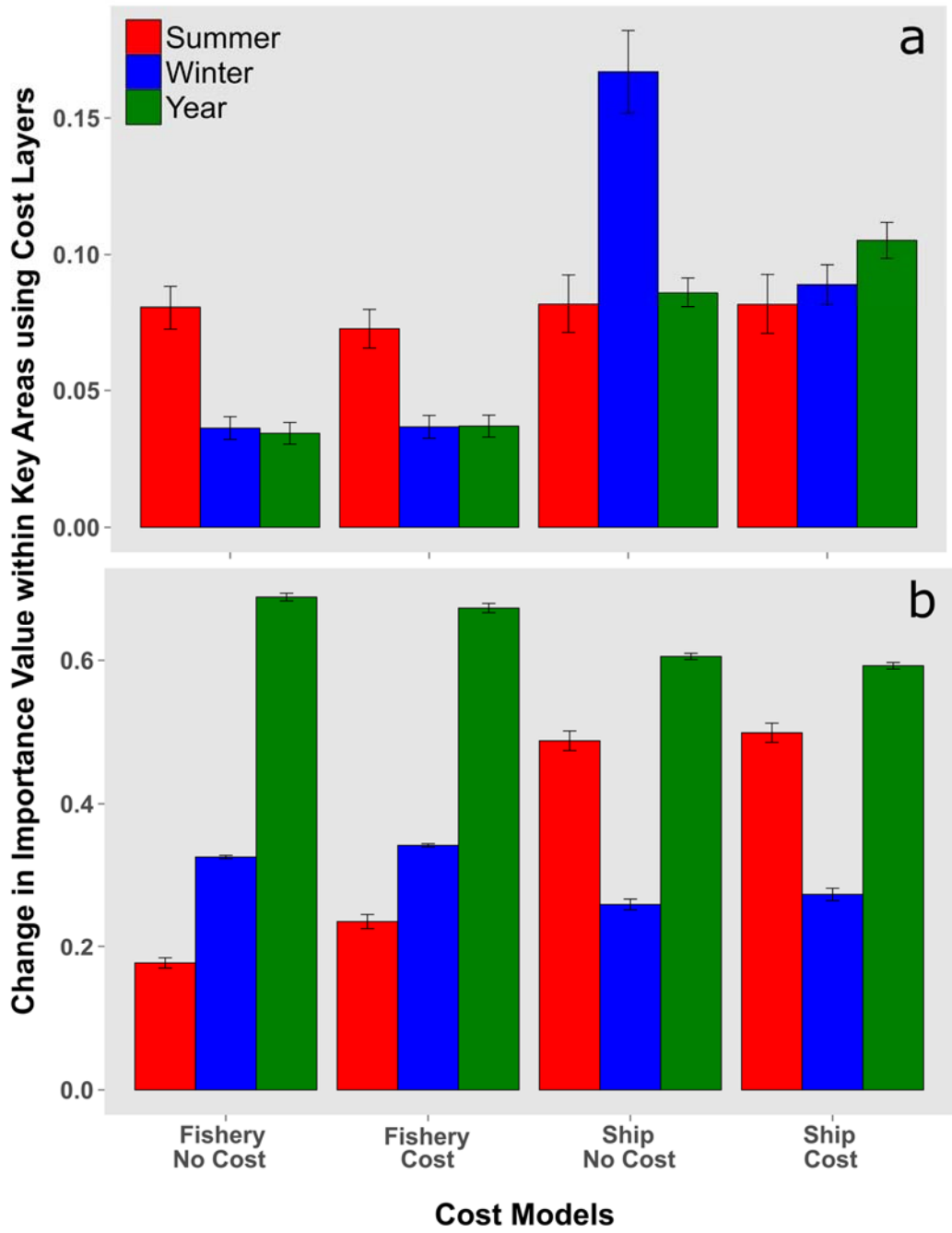


Area Values









Electronic Supplementary Material for Animal Conservation research article:

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

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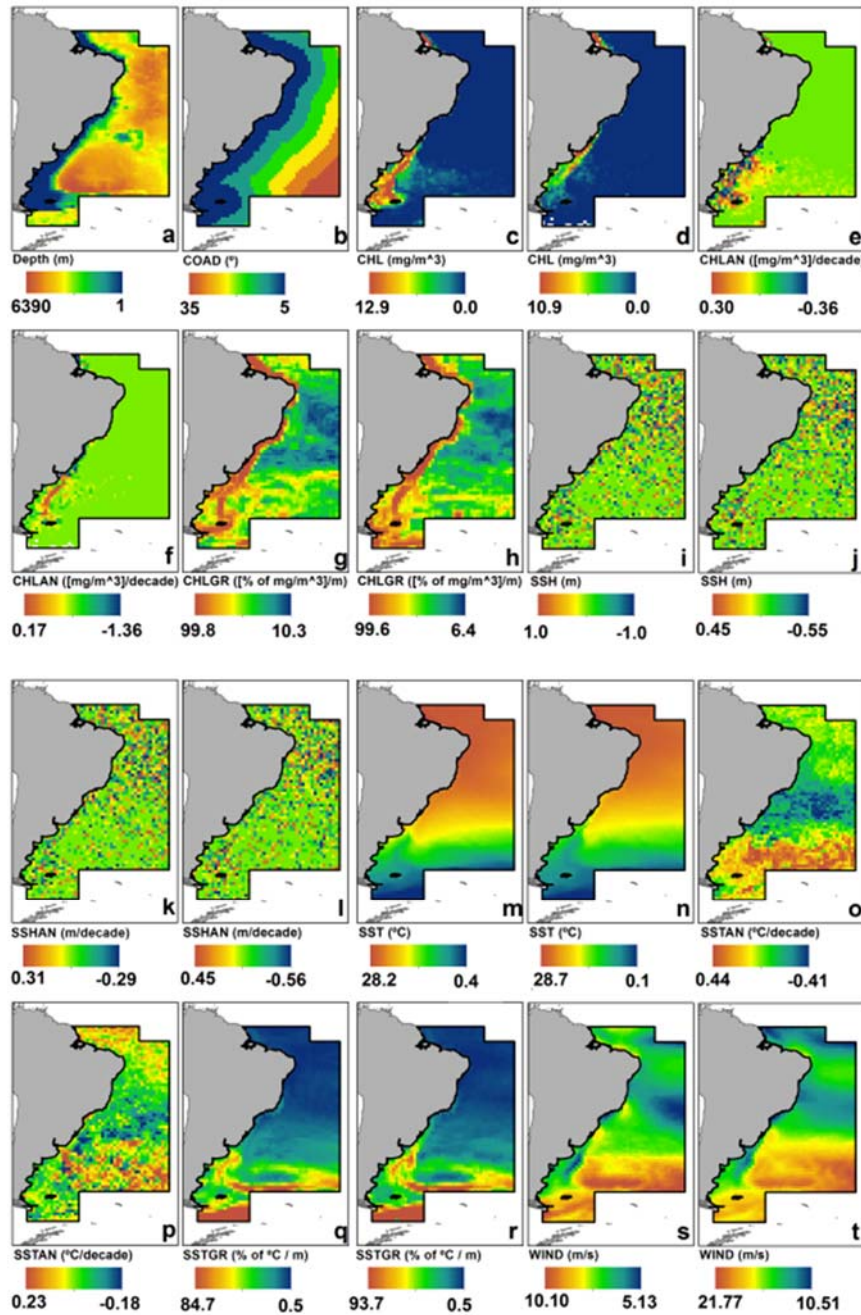


Figure S1 Environmental variables used for the Species Distribution Modelling: (a) depth in meters; (b) Distance to coast (COAD) in degrees; chlorophyll-*a* concentration (CHL) in mg/m^3 , during (c) summer and (d) winter; chlorophyll-*a* concentration anomaly CHLAN in decade variability of mg/m^3 , during (e) summer and (f) winter; CHL gradient (CHLGR) in percentage of mg/m^3 variation per meter during (g) summer and (h) winter; sea surface height SSH in meters during (i) summer and (j) winter; sea surface height anomaly (SSHAN) in variability of meters per decade during (k) summer and (l) winter; sea surface temperature SST in $^{\circ}\text{C}$ during (m) summer and (n) winter; sea surface temperature anomaly (SSTAN) in variability of $^{\circ}\text{C}$ per decade during (o) summer and (p) winter; sea surface temperature gradients in percentage of $^{\circ}\text{C}$ change per meter during (q) summer and (r) winter; wind speed in meters per second during (s) summer and (t) winter. Year variables were the average of both summer and winter.

SI Sensitivity analysis of the Zonation outputs when using different weighting values

The aim of this section is to provide an analysis of the method we used to weight species distribution by the IUCN conservation status, and the habitat connectivity response (in Zonation software called as *Boundary-Quality Penalty Curves BQPCs*). We used the full weight method that is in the main text to give value to the year-round distribution of all species.

Full model: 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, which are measures of species responses to habitat connection, were constructed empirically based on the IUCN conservation status. We generated linear responses with changing slopes for CR = -0.01, EN = -0.008, VU = -0.004, NT = -0.002 LC = 0.0.

We also generated three other different weight methods: a null model where the weight given for all species is the same (1) and the response to habitat connection (β^h) is constant; a model where LC weights 1 with β^h constant and the other levels (NT, VU, EN, CR) weights 2 with $\beta^h = -0.002$ (weight model #1), a model where LC weights 1 with β^h constant, NT weights 2 with $\beta^h = -0.002$, and other levels (VU, EN, CR) weight 3 with $\beta^h = -0.004$ (weight model #2). Then we tested via simple regression how these three methods are similar to the full model and tested how the probability of species occurrence is related to those weighting outputs through cubic regression.

We found that a null model with equal weight and no β^h variation means that the software will give more value to pixels isolated from the edges of the study area, without being realistic nor reflecting any oceanographic process intrinsic to the species distribution, but probably embracing the edges of most species distribution (Fig. S5). It means that high values in this case were concentrated in the middle of the study area (Fig. S5) and were completely opposed to the output we proposed ($F_{1,1652} = 3.29$, $\beta = -0.14$, $P=0.07$). On the other hand, one single differentiation on the weights in two ($F_{1,1652} = 31.18$, $\beta = 0.18$, $P<0.001$) and three ($F_{1,1652} = 1556.6$, $\beta = 0.8$, $P<0.001$) groups was enough to approximate the values to the full model output (Fig. S6). The null model weakly reflected the overall distribution of species ($R^2=0.04$, $F_{1,1652} = 24.55$, $P = 0.09$), while the other weight models #1 ($R^2 = 0.53$, $F_{1,1652} = 613.6$, $P<0.001$), #2 ($R^2 = 0.63$, $F_{1,1652} = 952.2$, $P<0.001$) and the full model ($R^2 = 0.69$, $F_{1,1652} = 1238.0$, $P<0.001$) did significantly explained the species distribution (Fig. S7).

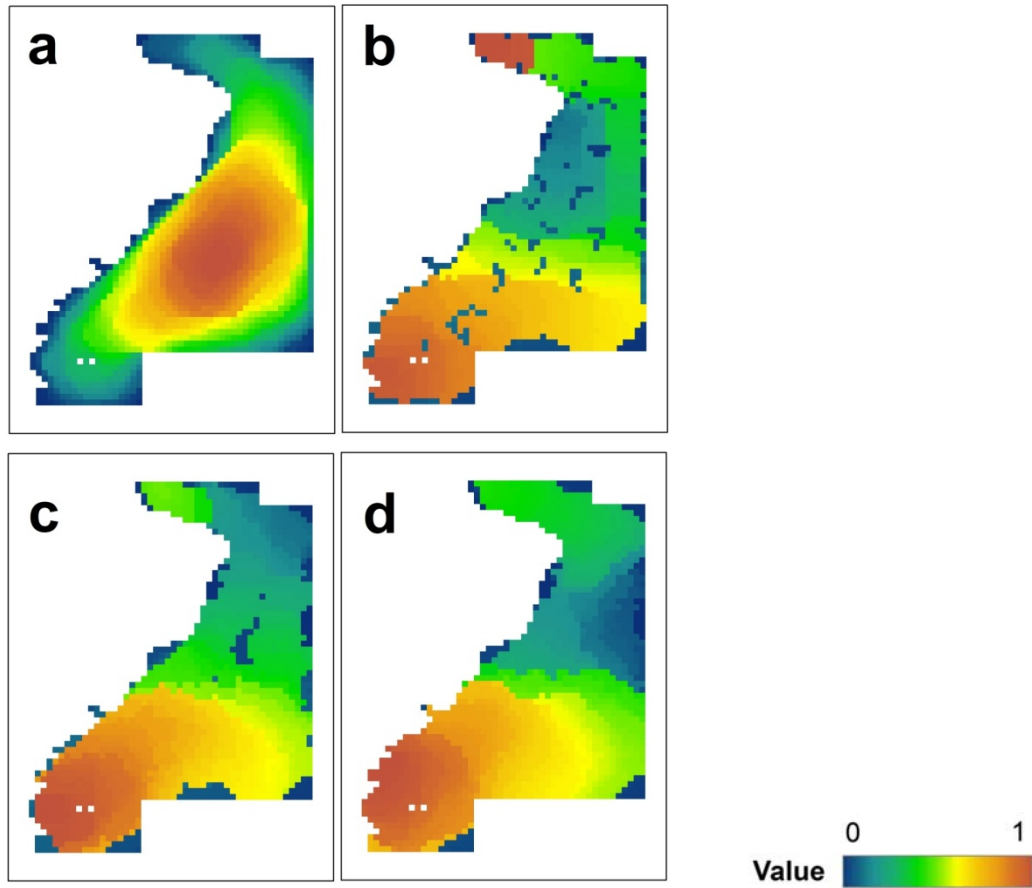


Figure S2 Zonation output showing the how much the values attributed to each pixel change by the weighting method used in building the (a) null model, (b) weight model #1, (c) weight model #2 and (d) the full model.

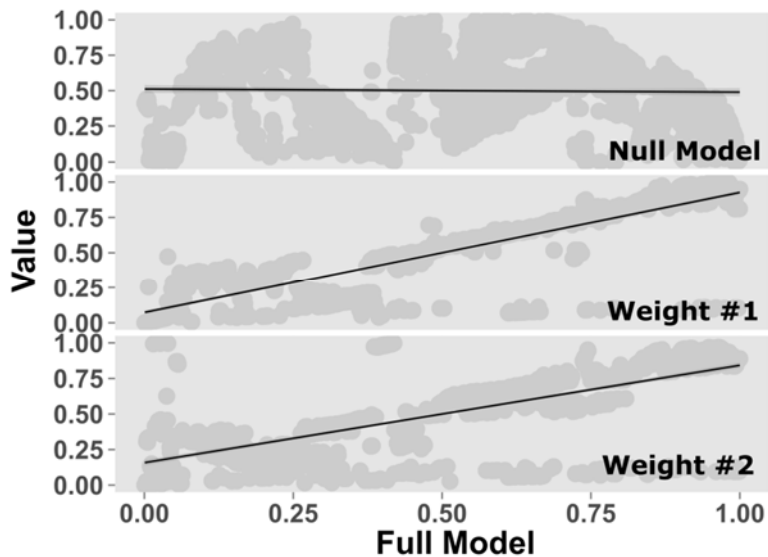


Figure S3 Regression between the full model and the new weight models. Lines are linear trends \pm SE.

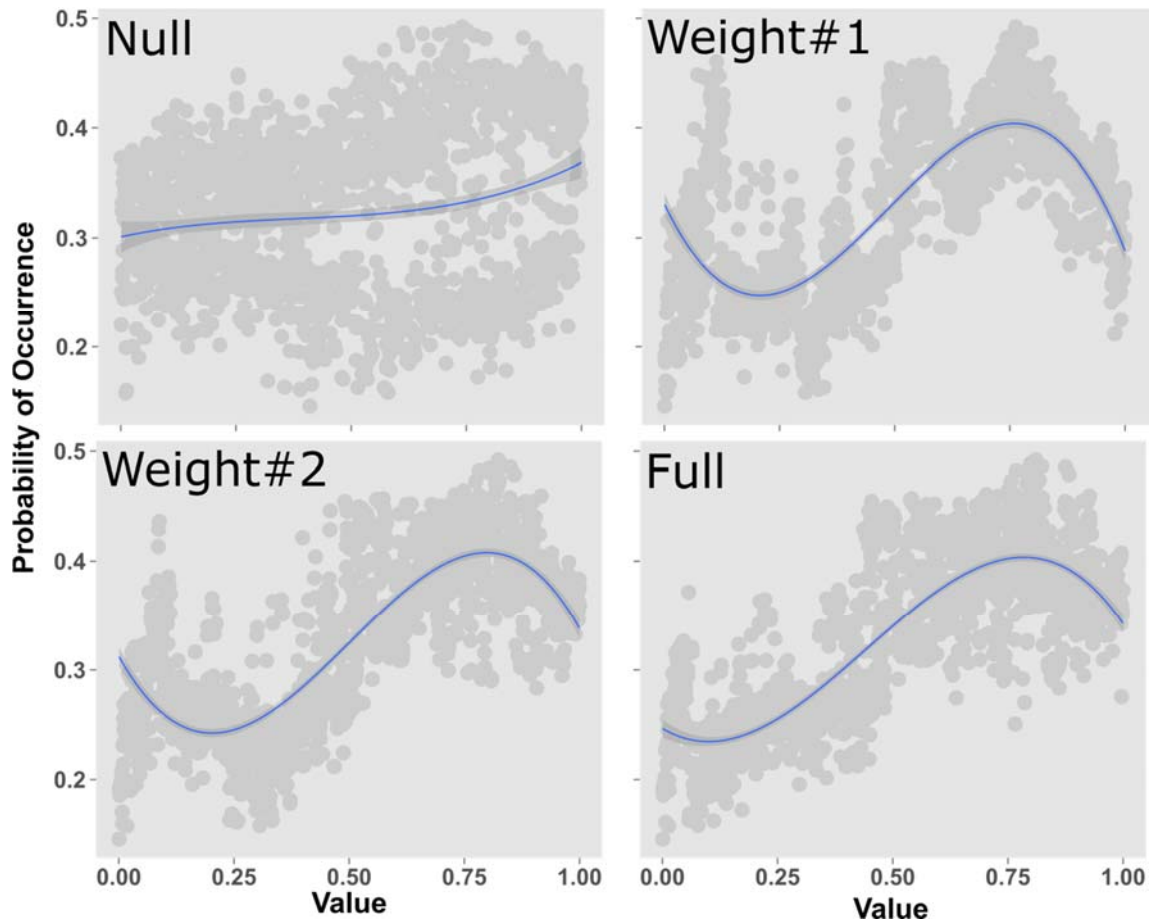


Figure S4 Relation between the overall species probability of occurrence and zonation value for each of the four models. Lines are cubic trend \pm SE.

SHI Methods for calculating environmental threat variables.

A total of five threat variables were used: ocean surface acidification, oil pollution risk, plastic accumulation, ship density and fishing intensity (Fig. S8). All environmental variables were standardized to have a mean of 0 and an SD of 1 due to differing ranges of variables. Such standardization was executed by dividing each absolute value by the respective maximal value of the environmental variable (Zuur, Ieno & Smith, 2007).

Ocean surface acidification was measured in terms of normalized changes in aragonite saturation state between pre-industrial and modern times, which is a compound that enhances when pH decreases (Halpern *et al.*, 2008). The acidification image was generated by Halpern *et al.* (2008), and we are using it as it is in the publication.

For risk of oil pollution we used the areas of oil accumulation proposed by Halpern *et al.* (2008). A Nearest Natural Neighbor Interpolation was used to calculate a risk based on the distance from these pollution areas, being the cells presented by Halpern *et al.* (2008) with value 1.0, and decreasing according to the distance.

Plastic accumulation was estimated using a dynamic particle flotation model based on movement of buoys (van Sebille, England & Froyland, 2012; van Sebille *et al.*, 2015; van Sebille, 2014). The model takes in account the coastal population density as a proxy for source of pollution, and predicts the probability of the movements of those particles due to ocean currents for each two month periods along a $1^\circ \times 1^\circ$ spatial grid. As there are different outputs if we use different periods of the year as the start of the modeling, we run models for all the two month periods along ten years. Each 2 month period output images were used to calculate an average image. This means that for cells with higher values there is a greater probability of plastic accumulation accounting for all possible scenarios. Thus the final images accounted also for the movement of the particles instead of only the final destination of the particles at the end of the 10-year period. We also considered a model with a constant particle release and a single release in time, to make the model more realistic, and calculated a mean of both.

Ship density was computed from the ship density shapefiles made available by the PASTA-MARE project (LuxSpace, 2010) which takes in account three month Satellite Automatic Identification System S-AIS movement of 62000 vessels on a global scale. The technical report (LuxSpace, 2010) shows a similarity between their results and two longer term data bank of the Voluntary Observing Ship from the World Meteorological Organization WMO-VOS (Halpern *et al.*, 2008) which uses one year data of 3374 commercial and research vessels, and Automated Mutual-assistance Vessel Rescue system AMVER that uses 3809 commercial vessels. We believe that the PASTA-MARE project is more accurate in terms of characterizing the ship density per area in relative terms because it uses all the available data for

any type and flag of vessels, while the more long-term data for VOS and AMVER are probably biased for the more representative types of vessels. We correlated the Halpern *et al.* (2008) with the PASTA-MARE results using a spatial correlation analysis on Spatial Analysis in Macroecology SAM software (Rangel, Diniz-Filho & Bini, 2010), and found a significant adjustment between both ($R=0.61$, $F=28.8$, $P<0.001$) meaning that the short term data from PASTA-MARE is a fair approximation for a year-long data bank. Fishing intensity was measured as the fishing vessel density from the PASTA-MARE data (LuxSpace, 2010).

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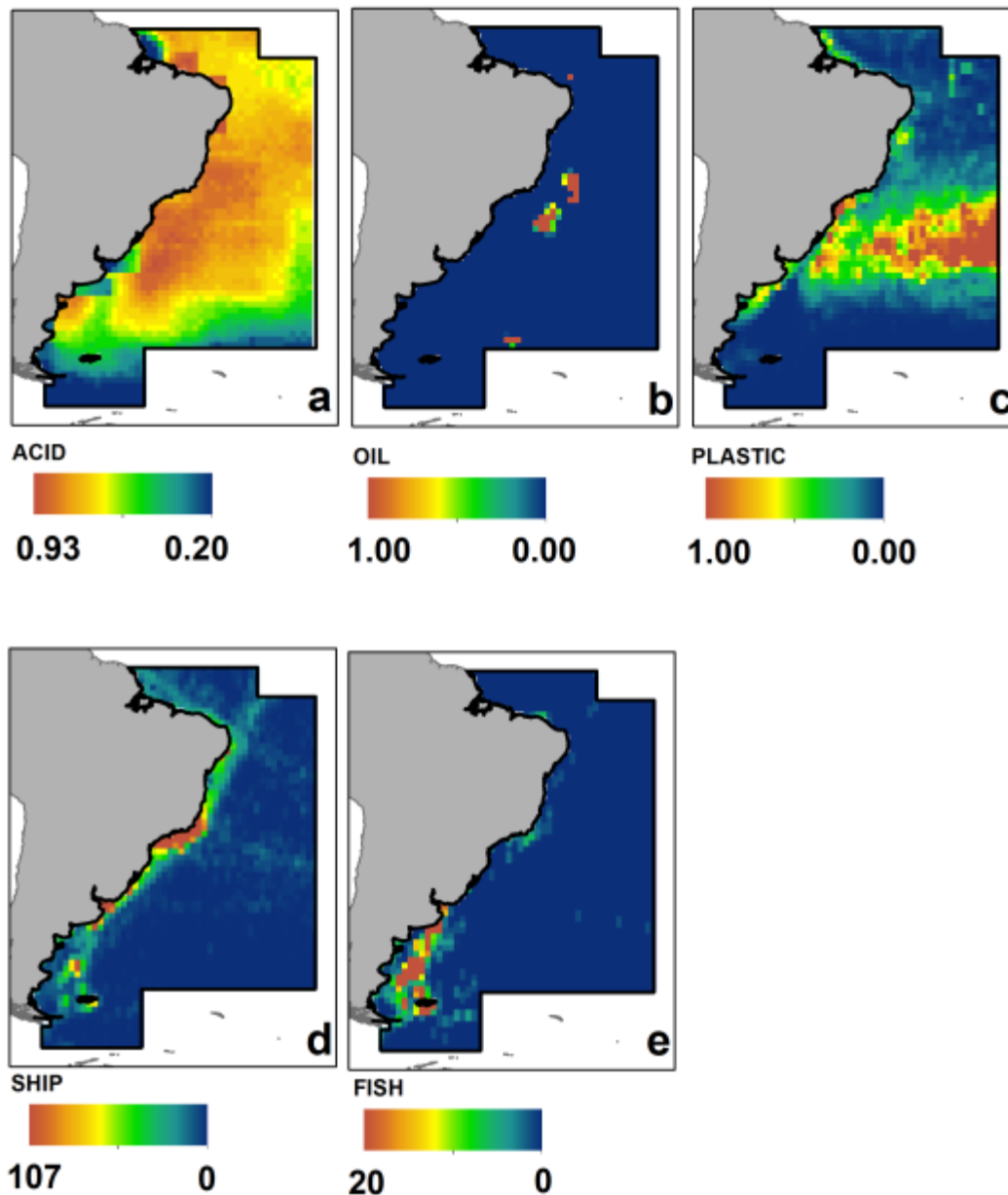


Figure S5 Environmental change variables: (a) ocean surface acidification (ACID) measured in terms of normalized changes in aragonite saturation state between pre-industrial and modern times (see Halpern et al 2008); (b) Risk of oil pollution (OIL) measured as a distance probability given the known points 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.

Table S1 Loadings from the PCA. The values represent the proportion of each variable that is explained by each of the PCA axis, and the signal (- or +) indicates the relation of the variables with the axes. Variables with higher correlation on the same axis are not independent from each other, meaning that they vary together. Table presents results from the axis whose eigenvalues were above 1.0. Higher values for each variable by season are presented in bold. Chlorophyll (CHL), CHL anomaly (CHLAN), CHL gradient (CHLGR), coast distance (COAD), Depth (DEPTH), Sea Surface Height (SSH), SSH anomaly (SSHAN), Sea Surface Temperature (SST), SST anomaly (SSTAN), SST gradient (SSTGR), wind speed (WIND).

Variables	Summer				Winter				Year			
	AX1	AX2	AX3	AX4	AX1	AX2	AX3	AX4	AX1	AX2	AX3	AX4
CHL	0.12	-0.64	0.10	-0.17	-0.03	0.60	0.00	0.57	0.01	-0.65	-0.06	-0.48
CHLAN	0.11	0.05	0.01	-0.98	0.09	-0.21	0.04	-0.77	0.12	0.22	0.01	0.85
CHLGR	0.43	-0.70	0.07	-0.01	0.57	0.67	0.06	-0.07	0.46	-0.74	-0.03	0.15
COAD	0.24	0.72	-0.03	0.02	0.25	-0.73	-0.10	0.34	0.31	0.70	0.03	-0.35
DEPTH	-0.13	0.87	-0.07	-0.08	-0.17	-0.83	-0.08	0.11	-0.08	0.86	0.04	-0.11
SSH	-0.03	0.09	0.99	0.01	-0.01	-0.09	0.99	0.05	-0.01	0.06	-0.95	0.00
SSHAN	-0.04	0.10	0.99	0.01	-0.01	-0.10	0.99	0.05	-0.01	0.07	-0.95	0.03
SST	-0.94	-0.03	-0.01	-0.08	-0.94	0.00	-0.02	0.05	-0.93	0.06	0.00	-0.02
SSTAN	0.75	0.16	0.01	-0.04	0.26	-0.16	-0.06	0.42	0.72	0.12	-0.03	-0.11
SSTGR	0.84	-0.03	0.00	0.09	0.85	0.08	0.01	-0.10	0.83	-0.12	0.01	0.09
WIND	0.83	0.23	0.00	0.00	0.78	-0.41	-0.03	0.10	0.82	0.29	0.01	-0.10

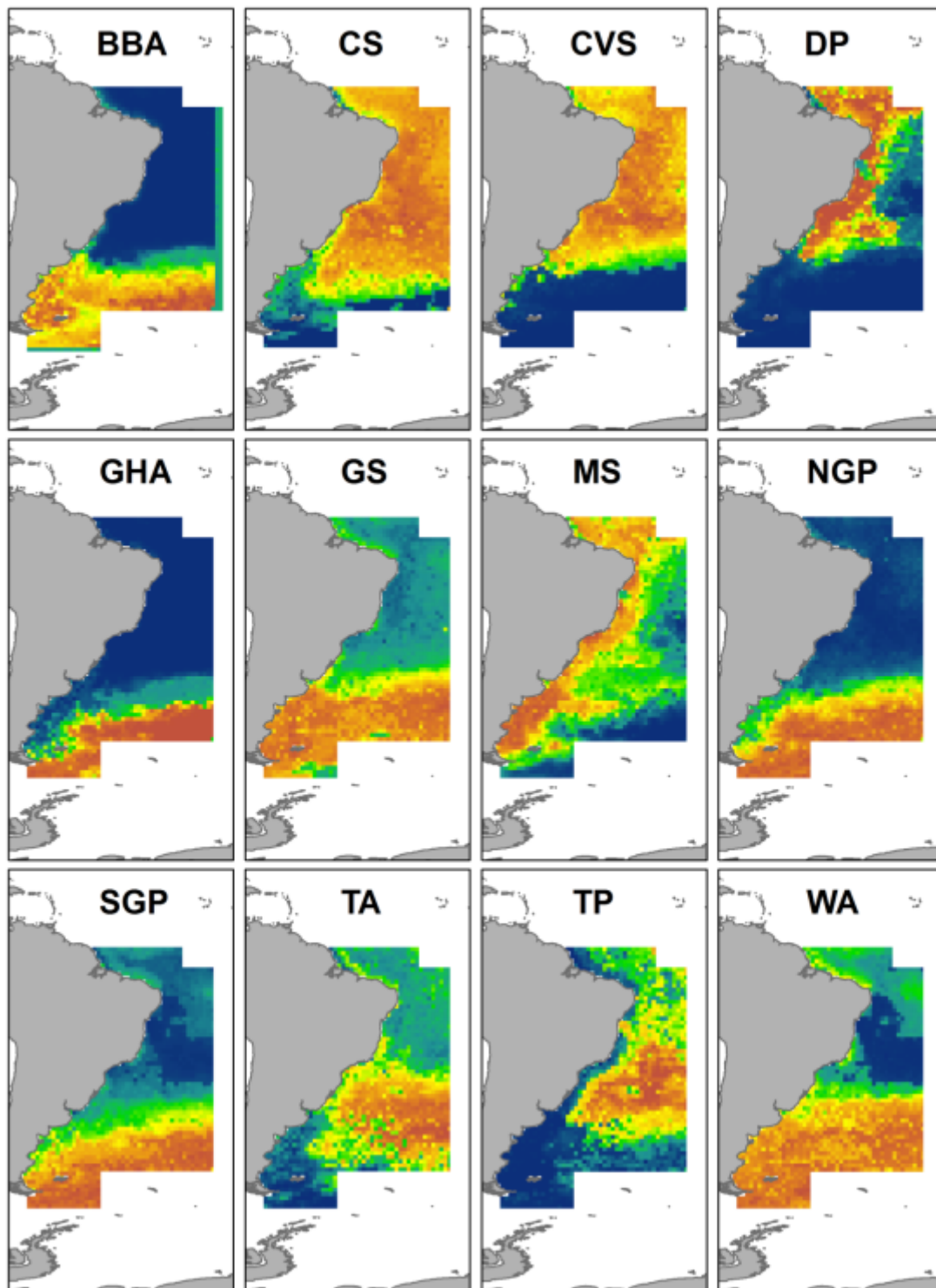


Figure S6 Summer probability of occurrence of Black-browed Albatross (BBA), Cory's Shearwater (CS), Cape Verde Shearwater (CVS), Deserta's Petrel (DP), Grey-headed Albatross (GHA), Great Shearwater (GS), Manx Shearwater (MS), Northern Giant Petrel (NGP), Southern Giant Petrel (SGP), Tristan Albatross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

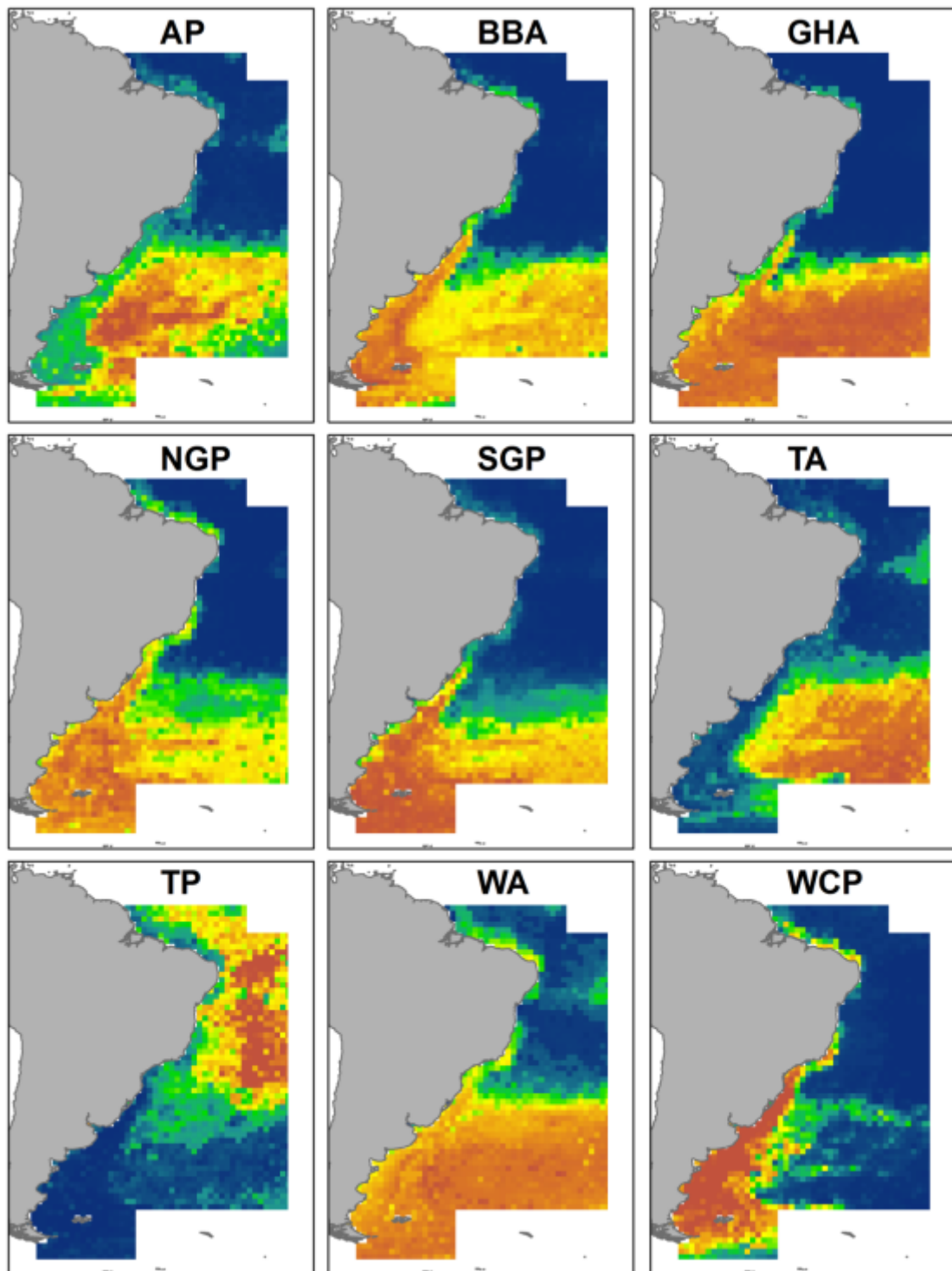


Figure S7 Winter probability of occurrence of Antarctic Prion (AP), Black-browed Albatross (BBA), Grey-headed Albatross (GHA), Northern Giant Petrel (NGP), Southern Giant Petrel (SGP), Tristan Albatross (TA), Trindade Petrel (TP), Wandering Albatross (WA), White-chinned Petrel (WCP), as estimated by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

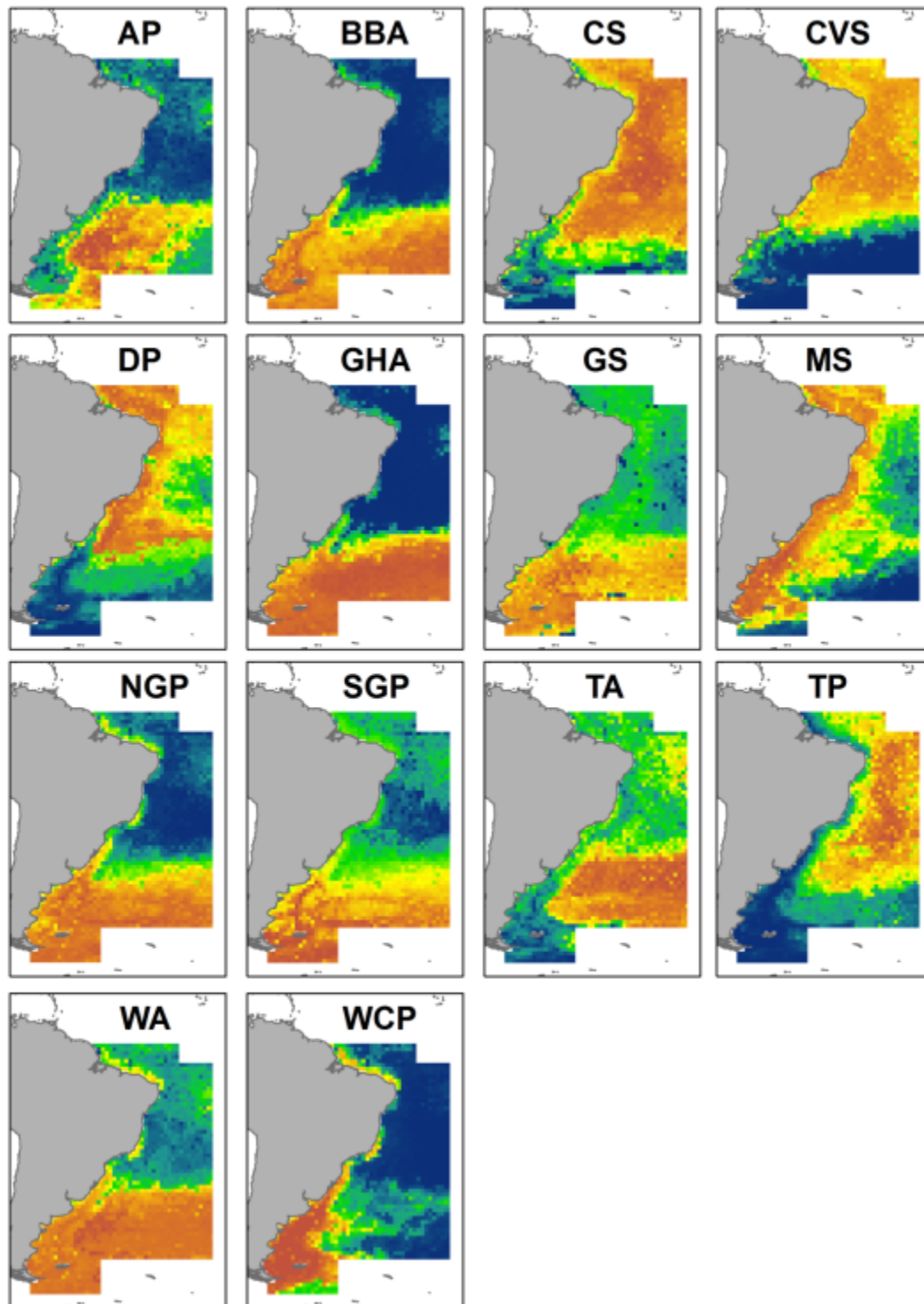


Figure S8 Year-round probability of occurrence of Antarctic Prion (AP), Black-browed Albatross (BBA), Cory's Shearwater (CS), Cape Verde Shearwater (CVS), Deserta's Petrel (DP), Grey-headed Albatross (GHA), Great Shearwater (GS), Manx Shearwater (MS), Northern Giant Petrel (NGP), Southern Giant Petrel (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).

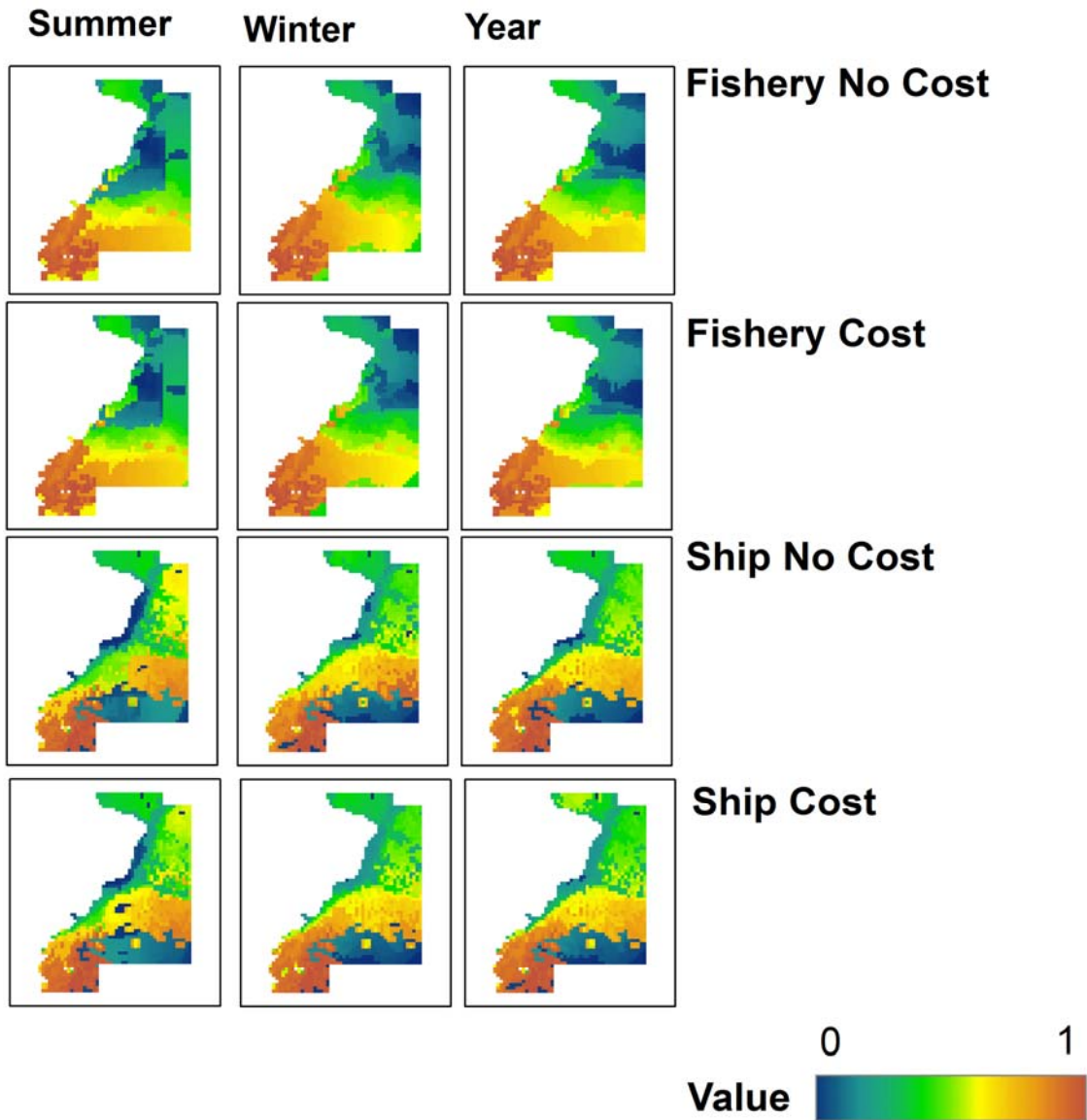


Figure S9 Zonation value for the different cost models, for southern seabirds.

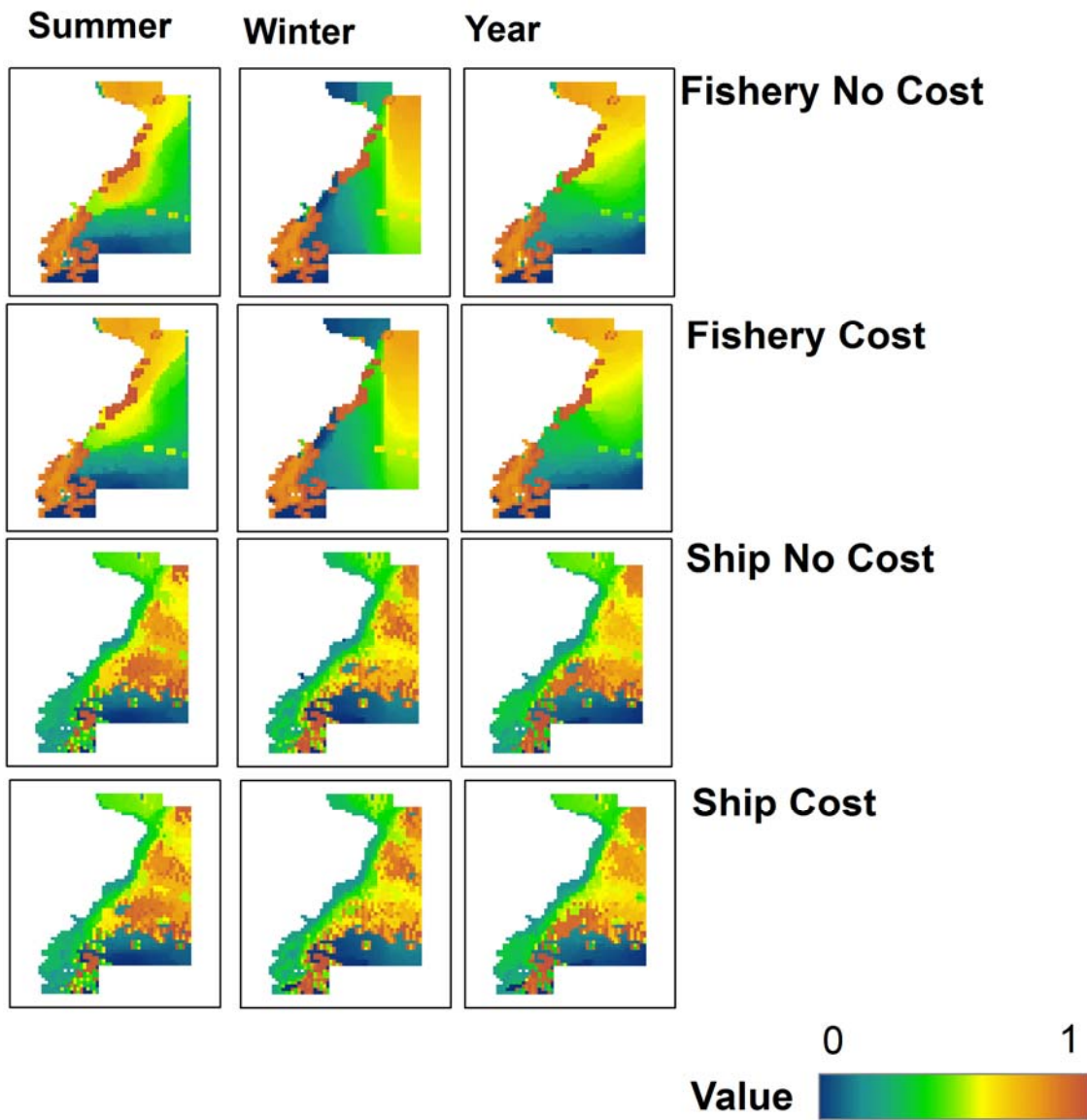


Figure S10 Zonation value for the different cost models models, for northern seabirds.

Table 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).

Models	β	SE	t
(Intercept)	0.0913	0.0142	6.44
Assemblage:Northern	0.0966	0.0113	8.58
Time-Frame:Winter	-0.0441	0.0113	-3.92
Time-Frame:Year	-0.0460	0.0113	-4.09
Cost-Model:Fishery Cost	-0.0078	0.0113	-0.7
Cost-Model:Ship No Cost	0.0015	0.0113	0.13
Cost-Model:Ship Cost	0.0014	0.0113	0.13
Assemblage:Northern* Time-Frame:Winter	0.1936	0.0162	11.93
Assemblage:Northern:* Time-Frame:Year	0.5585	0.0159	35.06
Assemblage:Northern* Cost-Model:Fishery Cost	0.0667	0.0159	4.19
Assemblage:Northern* Cost-Model:Ship No Cost	0.3094	0.0159	19.42
Assemblage:Northern* Cost-Model:Ship Cost	0.3206	0.0159	20.13
Time-Frame:Winter* Cost-Model:Fishery Cost	0.0083	0.0159	0.52
Time-Frame:Year* Cost-Model:Fishery Cost	0.0105	0.0159	0.66
Time-Frame:Winter* Cost-Model:Ship No Cost	0.1292	0.0159	8.11
Time-Frame:Year* Cost-Model:Ship No Cost	0.0502	0.0159	3.15
Time-Frame:Winter* Cost-Model:Ship Cost	0.0513	0.0159	3.22
Time-Frame:Year* Cost-Model:Ship Cost	0.0694	0.0159	4.36
Assemblage:Northern* Time-Frame:Winter* Cost-Model:Fishery Cost	-0.0509	0.0229	-2.22
Assemblage:Northern* Time-Frame:Year* Cost-Model:Fishery Cost	-0.0847	0.0225	-3.76
Assemblage:Northern* Time-Frame:Winter* Cost-Model:Ship No Cost	-0.5062	0.0229	-22.07
Assemblage:Northern* Time-Frame:Year* Cost-Model:Ship No Cost	-0.4450	0.0225	-19.76
Assemblage:Northern* Time-Frame:Winter* Cost-Model:Ship Cost	-0.4255	0.0229	-18.55
Assemblage:Northern* Time-Frame:Year* Cost-Model:Ship Cost	-0.4885	0.0225	-21.69